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Dynamac Corp., Rockville, MD

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16. ABSTRACT

This publication contains material from technology transfer seminars presented for the water supply community. The document provides utility managers and operators, regulators and technical specialists with guidance for preventing contamination and alternatives for treating public ground-water supplies that are contaminated.

An understanding of the processes that affect the movement and degradation of contaminants in the subsurface is essential for effective ground-water quality management. Knowledge can be gained from chapters on "Ground-Water Hydrology," "Classification of Ground-Water Regions" and Ground-Water--Surface Water Relationship."

Alternative management practices for dealing with contaminated ground-water supplies are discussed in chapters on "Ground-Water Pollution" and "Management Alternatives."

Available treatment options are discussed in chapters on "Controlling Volatile Organic Compounds in Ground Water Used for Drinking" and "Inground Treatment, Restoration and Reclamation."

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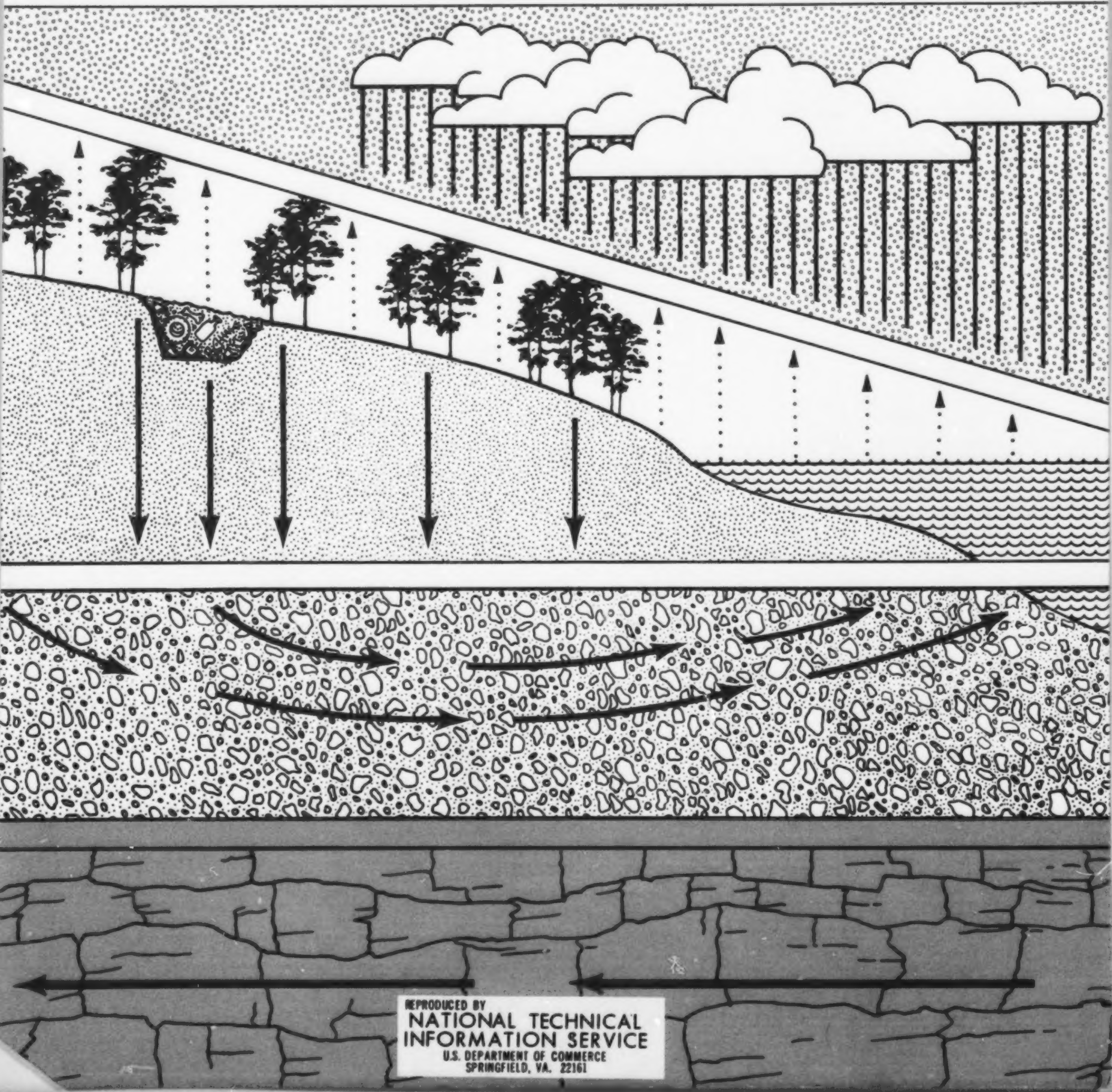

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This seminar publication contains material from technology transfer seminars for the water supply community. The U.S. Environmental Protection Agency's (EPA) Center for Environmental Research Information (CERI) and Office of Drinking Water sponsored 14 seminars, held in the 10 EPA Regions and in 4 states, to provide utility managers and operators, regulators, and technical specialists with procedures for preventing contamination and for treating public ground-water supplies that are contaminated. The seminars were designed and delivered in cooperation with the Water Engineering Research Laboratory in Cincinnati, Ohio, and the Robert S. Kerr Environmental Research Laboratory in Ada, Oklahoma; Regional and State coordinators arranged for presentations on local issues, case histories, and hydrogeologic considerations in dealing with ground-water contamination.

Though many individuals contributed to presenting the 14 seminars and to preparing and reviewing this publication, several deserve special recognition. Wayne A. Pettyjohn, Oklahoma State University, was principal speaker and author/editor for this project.

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Chapter 1

Introduction

Ground water is a valuable drinking water resource because of both availability and quality. In most locations of the country, it is available in quantities large enough to supply 50 percent of the nation's population. It also is a reliable resource because it is not subject to extremes in temperature or quality fluctuations and because it is buffered from the floods and droughts that affect surface supplies.

The development of subsurface water supplies has little effect on land use and can usually be accomplished at relatively low cost compared to the development of surface supplies. However, the subsurface environment is a complex system subject to contamination from a host of sources. Furthermore, the extremely slow movement of pollutants through this environment results in a longer residence time and little diffusion of many pollutants.

Because of the protracted effects of contamination and problems of accessibility, the restoration of ground-water quality is difficult and expensive. Restoration costs generally exceed the short-term value of the resource when compared to the costs of alternatives. For this reason, it is widely agreed that the most viable approach to ground-water quality protection is one of prevention rather than cure.

There will always be alternative management practices for obtaining drinking water from subsurface supplies. One may treat water withdrawn from the aquifer before use, protect the aquifer totally from all possible contaminant sources, or reclaim an aquifer after it has been comtaminated.

Existing Federal authority to address ground-water quality problems is embodied in at least eight statutes:

National Environmental Policy Act of 1970
Federal Water Pollution Control Act of 1972
Toxic Substances Control Act of 1976
Resource, Conservation and Recovery Act of 1976
Clean Water Act of 1977
Surface Mining Control and Reclamation Act of 1977
Safe Drinking Water Act of 1979
Comprehensive Environmental Response, Compensation and Liability Act of 1980

At the Federal level, the Safe Drinking Water Act, in particular, adds to the protection of ground water. In addition to establishing minimum drinking water standards, regardless of the source, it addresses the protection of ground-water quality and provides for research, technical assistance, and personnel training.

Protection of ground-water quality at the state level is gained from source control and from statutes dealing with surface water, which generally are based on public health concepts. Land-use regulations also are used to protect ground-water quality by locating waste sources, such as lagoons and solid waste facilities, in selected areas in order to minimize the pollution potential. Often, construction regulations are used in a secondary manner to protect ground water.

The most promising management option now available is to protect the ground-water resource from contamination. This will require many different "best management practices," including the development of protection plans at the local level to control activities that threaten the resource.

An understanding of the processes that affect the movement and degradation of contaminants in the subsurface is essential for effective ground-water quality management. The state of knowledge concerning these processes is, in many ways, insufficient to ensure protection of ground-water quality without excessive restrictions on other surface and subsurface activities.

It is evident from numerous case studies of contaminated water systems that local communities have been challenged by contamination of their public water supplies, and they have been able to find practical solutions. However, often the cost has been high in public anxiety and financial burden to the community.

Adequate prevention planning and emergency response may be beyond the resources of water utilities. Therefore, it becomes the responsibility of the local community to carry out planning and prevention programs that will assess drinking water needs and protect present and future water supplies.

While statutory authority exists to regulate most contamination sources there is a need for additional controls at the local level. Suitable aquifer protection controls can be adopted under local planning and zoning laws which take into account land use, industrial development, health, housing and agriculture. Different towns have different water supply needs and, thus, different ground-water protection needs. No one approach would successfully protect all aquifers. The entire community, including local government, is responsible for balancing the risks, costs and benefits involved in protectng the ground-water supply.

Community planning is particularly evident in some states, e.g., Connecticut, which has developed a nationally recognized ground-water program. The publication, "Protecting Connecticut's Ground Water," (available from Natural Resources Center, Connecticut Department of Environmental Protection, Hartford, Connecticut 06106) provides a guide for local officials on the issue of ground-water quality and possible mechanisms for local protection.

This publication provides an organized approach to acquiring the knowledge necessary for effective and efficient management of ground-water supplies. The information provided is applicable to all regions of the United States, taking into account differences in geographic location, from the humid to the arid, and in geologic composition, from the porous to the impermeable.

Chapter 2

Basic Ground-Water Hydrology

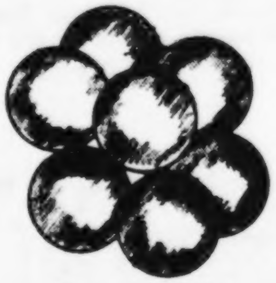
Introduction

The protection of ground water used for public supply is dependent, to a large extent, on an adequate understanding of the fundamentals of ground-water hydrology. Ground-water movement is neither mysterious nor occult but rather follows clearly defined principles of physics. The purpose of this chapter is to describe, in an accurate but simplified manner, (1) the physical characteristics of aquifers and how they function as storage media and as conduits, (2) techniques that are commonly applied in order to better understand fluid flow, and (3) methods that are used to determine or estimate aquifer coefficients.

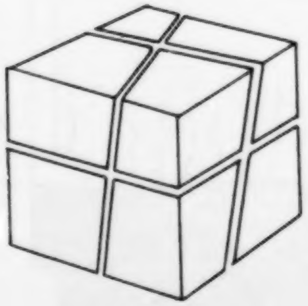
Rocks and Water

Most of the rocks near the Earth's surface are composed of both solids and voids, as Figure 1 shows. The solid part is, of course, much more obvious than the voids, but without the voids, there would be no water to supply wells and springs.

Water-bearing rocks consist either of unconsolidated (soil-like) deposits or consolidated rocks. The Earth's surface in most places is formed by soil and by unconsolidated deposits that range in thickness from a few centimeters near outcrops of consolidated rocks to more than 12,000 m beneath the delta of the Mississippi River. The unconsolidated deposits are underlain every-



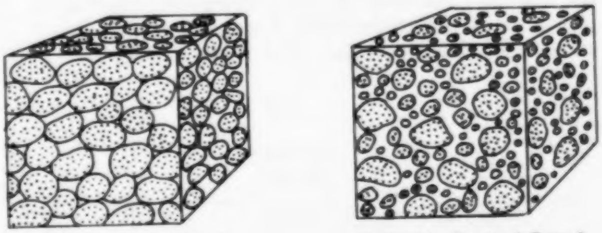
Porous Material



Fractured Rock

Figure 1. Composition of Rocks Near the Earth's Surface

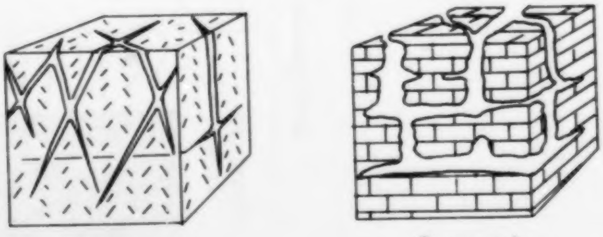
Primary Openings



Well-Sorted Sand

Poorly-Sorted Sand

Secondary Openings



Fractures in Granite

Caverns in Limestone

Figure 2. Different Kinds of Voids in Rocks

where by consolidated rocks.

Most *unconsolidated deposits* consist of material derived from the disintegration of consolidated rocks. The material consists, in different types of unconsolidated deposits, of particles of rocks or minerals ranging in size from fractions of a millimeter (clay size) to several meters (boulders). Unconsolidated deposits important in ground-water hydrology include, in order of increasing grain size, clay, silt, sand, and gravel. An important group of unconsolidated deposits also includes fragments of shells of marine organisms.

Consolidated rocks consist of mineral particles of different sizes and shapes that have been welded by heat and pressure or by chemical reactions into a solid mass. Such rocks are commonly referred to in ground-water reports as *bedrock*. They include sedimentary rocks that were originally unconsolidated and igneous rocks formed from a molten state. Consolidated sedimentary rocks important in ground-water hydrology include limestone, dolomite, shale, siltstone, sandstone, and conglomerate. Igneous rocks include granite and basalt.

Figure 2 shows that there are different kinds of voids in rocks, and it is sometimes useful to be aware of them. If the voids were formed at the same time as the rock, they are referred to as *primary openings*. The pores in sand and gravel and in other unconsolidated deposits are primary openings. The lava tubes and other openings in basalt are also primary openings.

If the voids were formed after the rock was formed, they are referred to as *secondary openings*. The fractures in granite and in consolidated sedimentary rocks are secondary openings. Voids in limestone, which are formed as ground water slowly dissolves the rock, are an especially important type of secondary opening.

It is useful to introduce the topic of rocks and water by dealing with unconsolidated deposits on one hand and with consolidated rocks on the other. It is important to note, however, that many sedimentary rocks that serve as sources of ground water fall between these extremes in a group of *semi-consolidated rocks*. These are rocks in which openings include both pores and fractures—in other words, both primary and secondary openings. Many limestones and sandstones that are important sources of ground water are semiconsolidated.

Underground Water

All water beneath the land surface is referred to as *underground water* (or subsurface water). The equivalent term for water on the land surface is *surface water*. As Figure 3 shows, underground water occurs in two different zones. One zone, which occurs immediately below the land surface in most areas, contains both water and air and is referred to as the *unsaturated zone*. The unsaturated zone is almost invariably underlain by a zone in which all interconnected openings are full of water. This zone is referred to as the *saturated zone*.

Water in the saturated zone is the only underground water that is available to supply wells and springs and is the only water to which the name *ground water* is correctly applied. Recharge of the saturated zone occurs by percolation of water from the land surface through the unsaturated zone. The unsaturated zone is, therefore, of great importance to ground-water hydrology. This zone may be divided usefully into three parts: the soil zone, the intermediate zone, and the upper part of the capillary fringe.

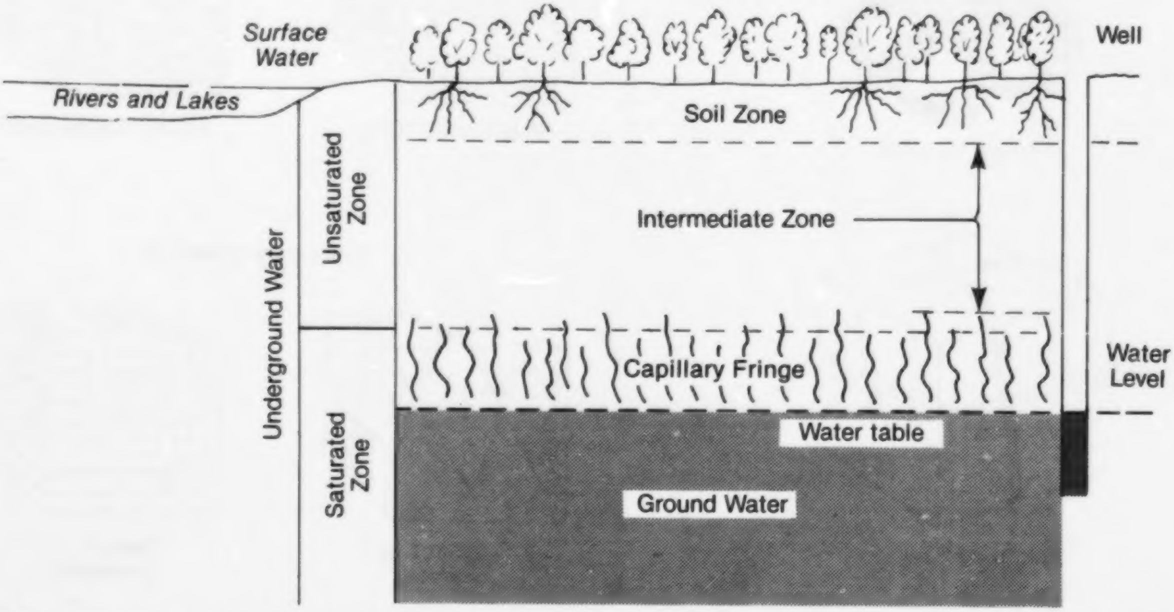


Figure 3. Underground Water Zones

The soil zone extends from the land surface to a maximum depth of a meter or two and is the zone that supports plant growth. It is crisscrossed by living roots, by voids left by decayed roots of earlier vegetation, and by animal and worm burrows. The porosity and permeability of this zone tend to be higher than those of the underlying material. The soil zone is underlain by the *intermediate zone,* which differs in thickness from place to place depending on the thickness of the soil zone and the depth to the capillary fringe.

The lowest part of the unsaturated zone is occupied by the *capillary fringe,* the subzone between the unsaturated and saturated zones. The capillary fringe results from the attraction between water and rocks. As a result of this attraction, water clings as a film on the surface of rock particles and rises in small-diameter pores against the pull of gravity. Water in the capillary fringe and in the overlying part of the unsaturated zone is under a negative hydraulic pressure—that is, it is under a pressure less than the atmospheric (barometric) pressure. The *water table* is the level in the saturated zone at which the hydraulic pressure is equal to atmospheric pressure and is represented by the water level in unused wells. Below the water table, the hydraulic pressure increases with increasing depth.

Hydrologic Cycle

The term *hydrologic cycle* refers to the constant movement of water above, on, and below the Earth's surface. Figure 4 illustrates this cycle. The concept of the hydrologic cycle is central to an understanding of the occurrence of water and the development and management of water supplies.

Although the hydrologic cycle has neither a beginning nor an end, it is convenient to discuss its principal features by starting with evaporation from vegetation, from exposed moist surfaces including the land surface, and from the ocean. This moisture forms clouds, which return the water to the land surface or oceans in the form of precipitation.

Precipitation occurs in several forms, including rain, snow, and hail, but only rain is considered in this discussion. The first rain wets vegetation and other surfaces and then begins to infiltrate into the ground. *Infiltration* rates vary widely, depending on land use, the character and moisture content of the soil, and the intensity and duration of precipitation, from possibly as much as 25 mm/hr in mature forests on sandy soils to a few millimeters per hour in clayey and silty soils to zero in paved areas. When and if the rate of precipitation exceeds the rate of infiltration, *overland flow* occurs.

The first infiltration replaces soil moisture, and thereafter, the excess percolates slowly across the intermediate zone to the zone of saturation. Water in the zone of saturation moves downward and laterally to sites of ground-water discharge such as springs on hillsides or seeps in the bottoms of streams and lakes or beneath the ocean.

Water reaching streams, both by overland flow and from ground-water discharge, moves to the sea, where it is again evaporated to perpetuate the cycle.

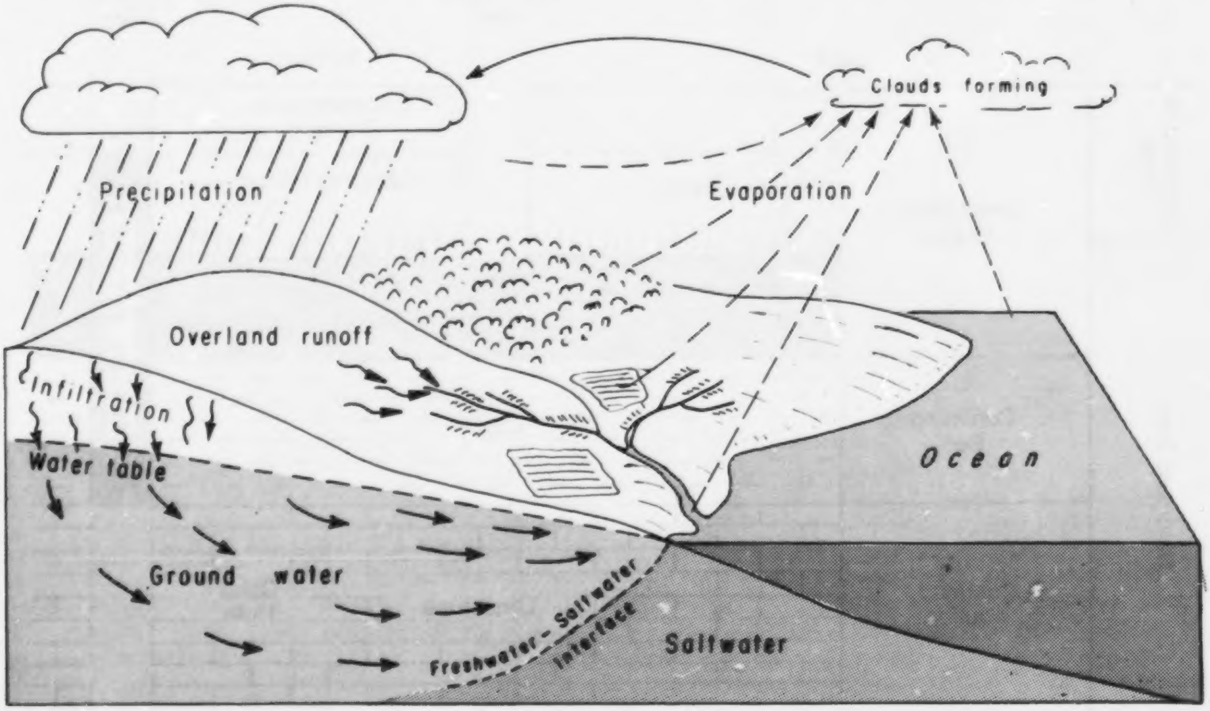


Figure 4. The Hydrologic Cycle

Movement is, of course, the key element in the concept of the hydrologic cycle. Table 1 shows some "typical" rates of movement along with the distribution of the Earth's water supply.

Table 1. Rate of Movement and Distribution of Water
[Adapted from L'vovich (1979), table 1]

Location	Rate of Movement	Distribution of Earth's Water Supply (percent)
Atmosphere	100's of kilometers per day	0.001
Water on land surface	10's of kilometers per day	.019
Water below the land surface	Meters per year	4.12
Ice caps and glaciers	Meters per day	1.65
Oceans	—	93.96

Aquifers and Confining Beds

From the standpoint of ground-water occurrence, all rocks that underlie the Earth's surface can be classified either as aquifers or as confining beds. Figure 5 illustrates these formations. An *aquifer* is a rock unit that will yield water in a usable quantity to a well or spring. (In geologic usage, "rock" includes unconsolidated sediments.) A *confining bed* is a rock unit having very low hydraulic conductivity that restricts the movement of ground water either into or out of adjacent aquifers.

Ground water occurs in aquifers under two different conditions. Where water only partly fills an aquifer, the upper surface of the saturated zone is free to rise and decline. The water in such aquifers is said to be *unconfined,* and the aquifers are referred to as *unconfined aquifers.* Unconfined aquifers are also widely referred to as water-table aquifers.

Where water completely fills an aquifer that is overlain by a confining bed, the water in the aquifer is said to be *confined.* Such aquifers are referred to as *confined aquifers* or as *artesian aquifers.*

Wells open to unconfined aquifers are referred to as *water-table wells.* The water level in these wells indicates the position of the water table in the surrounding aquifer.

Wells drilled into confined aquifers are referred to as *artesian wells.* The water level in artesian wells stands at some height above the top of the aquifer but not necessarily above the land surface. If the water level in an artesian well stands above the land surface, the well is a *flowing artesian well.* The water level in tightly cased wells open to a confined aquifer stands at the level of the *potentiometric surface* of the aquifer.

Porosity

The ratio of openings (voids) to the total volume of a soil or rock is referred to as its *porosity.* Porosity is ex-

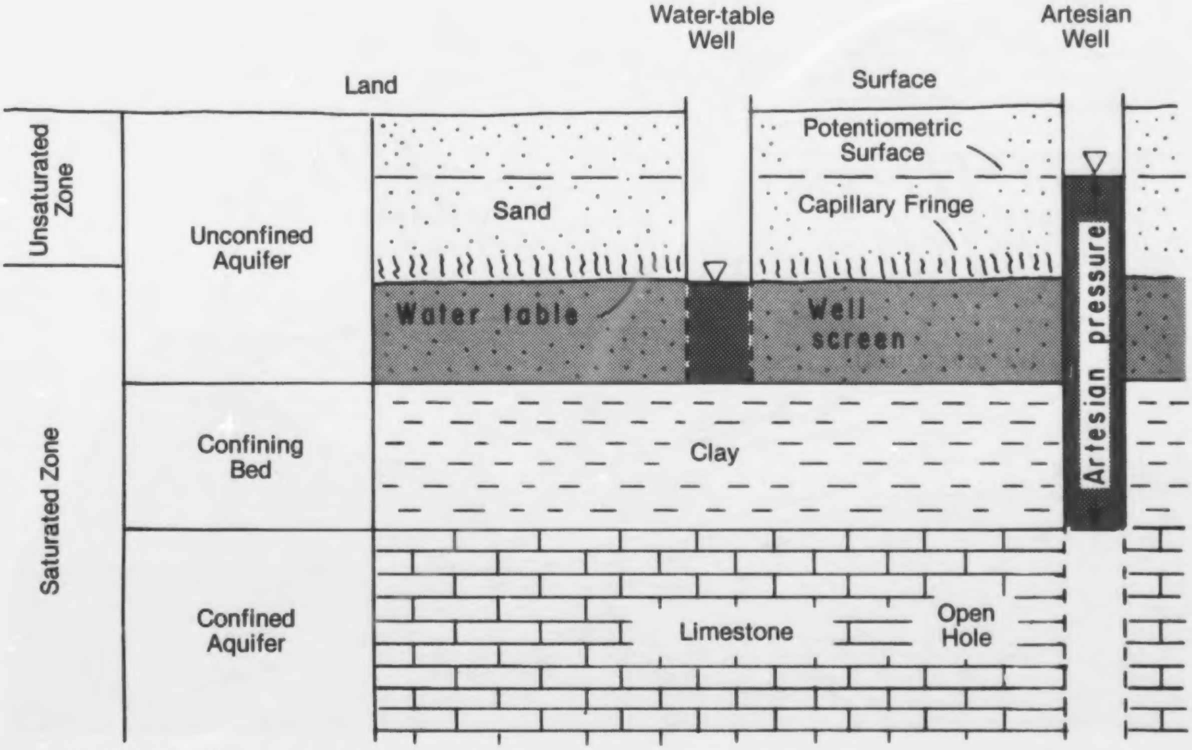


Figure 5. Aquifers and Confining Bed

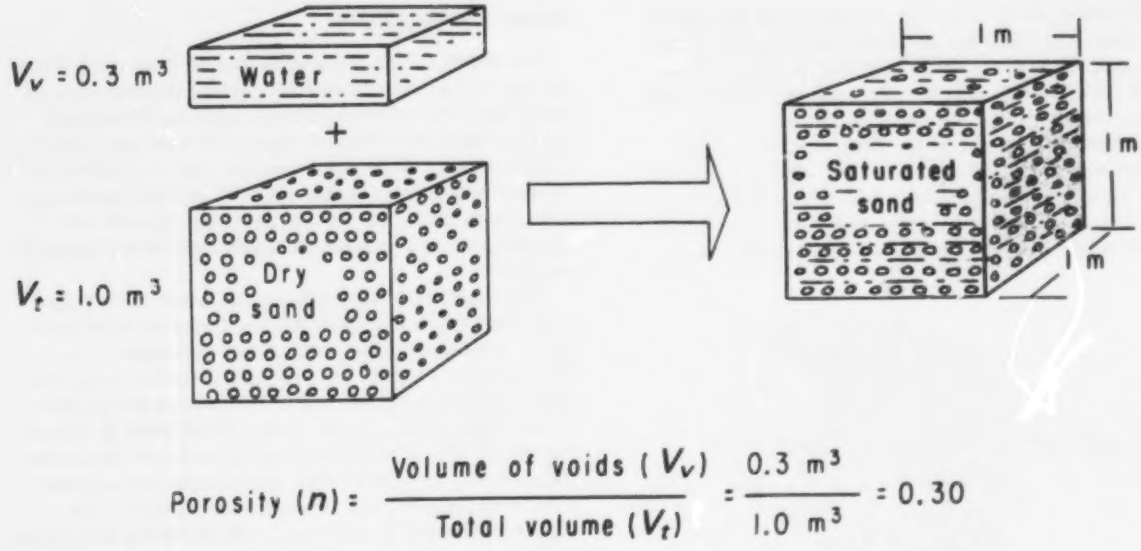

V_v = 0.3 m³
Water
+
V_t = 1.0 m³
Dry sand
1 m
1 m
1 m
Saturated sand
Porosity (n) = Volume of voids (V_v) / Total volume (V_t) = 0.3 m³ / 1.0 m³ = 0.30

Figure 6. Illustration of Porosity

pressed either as a decimal fraction or as a percentage. Thus,

$$n = \frac{V_t - V_s}{V_t} = \frac{V_v}{V_t} \qquad (1)$$

where n is porosity as a decimal fraction, V_t is the total volume of a soil or rock sample, V_s is the volume of solids in the sample, and V_v is the volume of openings (voids). Figure 6 illustrates the calculation of porosity.

If we multiply the porosity determined with the equation by 100, the result is porosity expressed as a percentage.

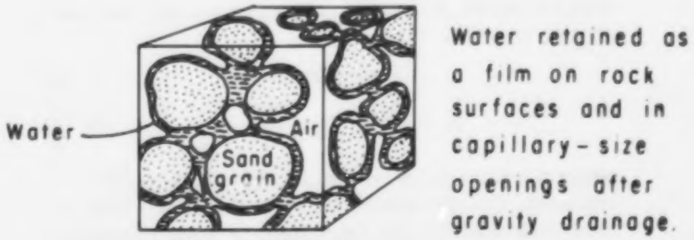
Table 2 lists porosity values for some selected materials. Soils are among the most porous of natural materials because soil particles tend to form loose clumps and because of the presence of root holes and animal burrows. Porosity of unconsolidated deposits depends on the range in grain size (sorting) and on the shape of the rock particles but not on their size. Fine-grained materials tend to be better sorted and, thus, tend to have the largest porosities.

Table 2. Selected Values of Porosity
[Values in percent by volume]

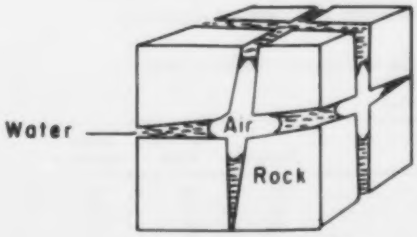
Material	Primary Openings	Secondary Openings
Equal-size spheres (marbles):		
Loosest packing	48	—
Tightest packing	26	—
Soil	55	—
Clay	50	—
Sand	25	—
Gravel	20	—
Limestone	10	10
Sandstone (semiconsolidated)	10	1
Granite	—	.1
Basalt (young)	10	1

Specific Yield and Specific Retention

Porosity is important in ground-water hydrology because it tells us the maximum amount of water that a rock can contain when it is saturated. However, it is equally important to know that only a part of this water is available to supply a well or a spring. Figure 7 gives some graphical examples of porosity.


Water
Air
Sand grain
Water retained as a film on rock surfaces and in capillary-size openings after gravity drainage.

Granular Material



Fractured Rock

Figure 7. Graphical Example of Porosity

Hydrologists divide water in storage in the ground into the part that will drain under the influence of gravity (called *specific yield*) and the part that is retained as a film on rock surfaces and in very small openings (called *specific retention*). The physical forces that control specific retention are the same forces involved in the thickness and moisture content of the capillary fringe.

Specific yield tells how much water is available for man's use, and specific retention tells how much water remains in the rock after it is drained by gravity. As Figure 8 shows, porosity is the sum of specific yield and specific retention. Thus,

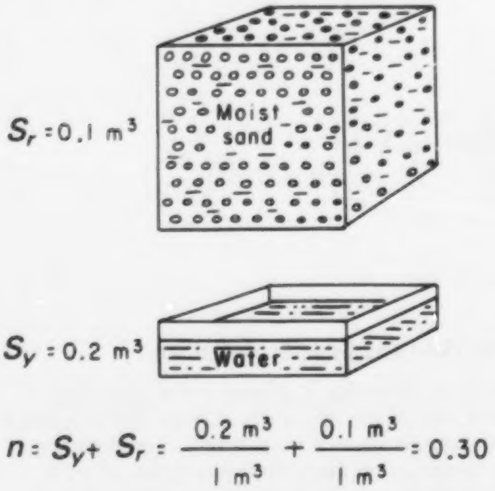

$S_r = 0.1\ m^3$
Moist sand
$S_y = 0.2\ m^3$
Water
$n = S_y + S_r = \frac{0.2\ m^3}{1\ m^3} + \frac{0.1\ m^3}{1\ m^3} = 0.30$

Figure 8. Porosity is the Sum of Specific Yield and Specific Retention

$$n = S_y + S_r \tag{2}$$

$$S_y = \frac{V_d}{V_t} \qquad S_r = \frac{V_r}{V_t} \qquad (3), (4)$$

where n is porosity, S_y is specific yield, S_r is specific retention, V_d is the volume of water that drains from a total volume of V_t, V_r is the volume of water retained in a total volume of V_t, and V_t is total volume of a soil or rock sample. Table 3 lists values of porosity, specific yield, and specific retention for selected materials.

Table 3. Selected Values of Porosity, Specific Yield, and Specific Retention
[Values in percent by volume]

Material	Porosity	Specific Yield	Specific Retention
Soil	55	40	15
Clay	50	2	48
Sand	25	22	3
Gravel	20	19	1
Limestone	20	18	2
Sandstone (semiconsolidated)	11	6	5
Granite	.1	.09	.01
Basalt (young)	11	8	3

Heads and Gradients

The depth to the water table has an important effect on use of the land surface and on the development of water supplies from unconfined aquifers. Where the water table is at a shallow depth, the land may become "waterlogged" during wet weather and unsuitable for residential and many other uses. Where the water table is at great depth, the cost of constructing wells and pumping water for domestic needs may be prohibitively expensive.

The direction of the slope of the water table is also important because it indicates the direction of ground-water movement. The position and the slope of the water table (or of the potentiometric surface of a confined aquifer) is determined by measuring the position of the water level in wells from a fixed point (a measuring point). To utilize these measurements to determine the slope of the water table, the position of the water table at each well must be determined relative to a *datum plane* that is common to all the wells. The datum plane most widely used is the National Geodetic Vertical Datum of 1929 (also commonly referred to as "sea level").

If the depth to water in a nonflowing well is subtracted from the altitude of the measuring point, the result is the *total head* at the well. Total head, as defined in fluid mechanics, is composed of *elevation head, pressure head,* and *velocity head.* Because ground water moves relatively slowly, velocity head can be ignored. Therefore, the total head at an observation well involves only two components: elevation head and pressure head. Ground water moves in the direction of decreasing total head, which may or may not be in the direction of decreasing pressure head.

The equation for total head (h_t) is

$$h_t = z + h_p \tag{5}$$

where z is elevation head and is the distance from the datum plane to the point where the pressure head h_p is determined.

All other factors being constant, the rate of ground-water movement depends on the *hydraulic gradient.* The hydraulic gradient is the change in head per unit of distance in a given direction. If the direction is not specified, it is understood to be in the direction in which the maximum rate of decrease in head occurs.

If the movement of ground water is assumed to be in the plane of Figure 9—in other words, if it moves from well 1 to well 2—the hydraulic gradient can be calculated from the information given on the drawing. The hydraulic gradient is h_L/L, where h_L is the head loss between wells 1 and 2 and L is the horizontal distance between them, or

$$\frac{h_L}{L} = \frac{(100\ m - 15\ m) - (98\ m - 18\ m)}{780\ m} = \frac{85\ m - 80\ m}{780\ m} = \frac{5\ m}{780\ m}$$

When the hydraulic gradient is expressed in consistent units, as it is in the above example in which both the numerator and the denominator are in meters, any

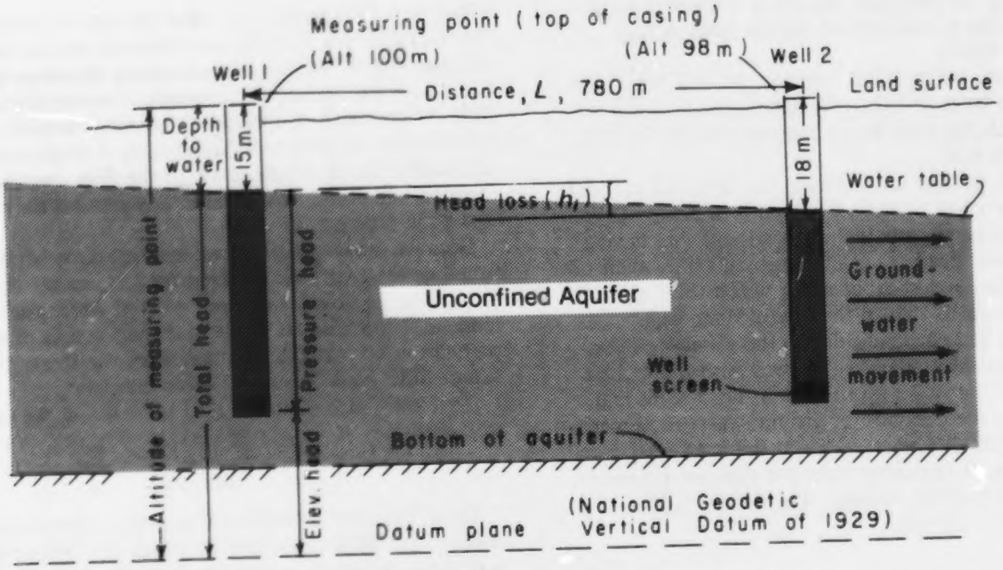


Figure 9. Gradient is Determined By the Difference in Head Between Two Wells.

other consistent units of length can be substituted without changing the value of the gradient. Thus, a gradient of 5 ft/780 ft is the same as a gradient of 5m/780 m. It is also relatively common to express hydraulic gradients in inconsistent units such as meters per kilometer or feet per mile. A gradient of 5 m/780 m can be converted to meters per kilometer as follows:

$$\left(\frac{5\,\text{m}}{780\,\text{m}}\right) \times \left(\frac{1{,}000\,\text{m}}{\text{km}}\right) = 6.4\,\text{m}\,\text{km}^{-1}$$

Both the direction of ground-water movement and the hydraulic gradient can be determined if the following data are available for three wells located in any triangular arrangement such as that shown in Figure 10:

1. The relative geographic position of the wells.
2. The distance between the wells.
3. The total head at each well.

Figure 11 illustrates the following steps in the solution.

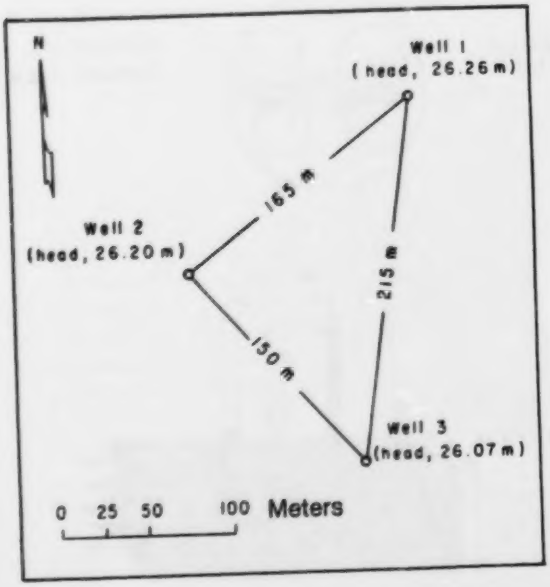


Figure 10. Triangular Arrangement of Wells

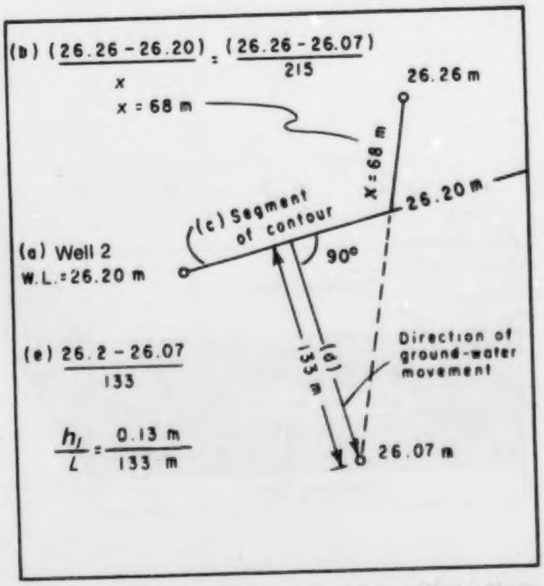

(b) $\frac{(26.26-26.20)}{x} = \frac{(26.26-26.07)}{215}$
$x = 68$ m
26.26 m
$x = 68$ m
26.20 m
(c) Segment of contour
(a) Well 2
W.L.=26.20 m
90°
(e) $\frac{26.2-26.07}{133}$
$\frac{h_l}{L} = \frac{0.13\text{ m}}{133\text{ m}}$
(d) 133 m
Direction of ground-water movement
26.07 m

Figure 11. Determining the Direction of Ground-Water Movement and the Hydraulic Gradient for a Triangular Arrangement of Wells

a. Identify the well that has the intermediate water level (that is, neither the highest head nor the lowest head).
b. Calculate the position between the well having the highest head and the well having the lowest head at which the head is the same as that in the intermediate well.
c. Draw a straight line between the intermediate well and the point identified in step b as being between the well having the highest head and that having the lowest head. This line represents a segment of the water-level contour along which the total head is the same as that in the intermediate well.
d. Draw a line perpendicular to the water-level contour and through either the well with the highest head or the well with the lowest head. This line parallels the direction of ground-water movement.
e. Divide the difference between the head of the well and that of the contour by the distance between the well and the contour. The answer is the hydraulic gradient.

Hydraulic Conductivity

Aquifers transmit water from recharge areas to discharge areas and thus function as porous conduits (or pipelines filled with sand or other water-bearing material). The factors controlling ground-water movement were first expressed in the form of an equation by Henry Darcy, a French engineer, in 1856. Darcy's law is

$$Q = KA\left(\frac{dh}{dl}\right) \qquad (6)$$

where Q is the quantity of water per unit of time; K is the hydraulic conductivity and depends on the size and arrangement of the water-transmitting openings (pores and fractures) and on the dynamic characteristics of the fluid (water) such as kinematic viscosity, density, and the strength of the gravitational field; A is the cross-sectional area, at a right angle to the flow direction, through which the flow occurs; and dh/dl is the hydraulic gradient.[1]

Because the quantity of water (Q) is directly proportional to the hydraulic gradient (dh/dl), we say that ground-water flow is *laminar*—that is, water particles tend to follow discrete streamlines and not to mix with particles in adjacent streamlines. (See the "Ground-Water Flow Nets" section of this chapter.)

If we rearrange equation 6 to solve for K, we obtain

$$K = \frac{Qdl}{Adh} = \frac{(\mathrm{m^3 d^{-1}})(\mathrm{m})}{(\mathrm{m^2})(\mathrm{m})} = \frac{\mathrm{m}}{\mathrm{d}} \qquad (7)$$

Thus, the units of hydraulic conductivity are those of velocity (or distance divided by time). It is important to note from equation 7, however, that the factors involved in the definition of hydraulic conductivity include the volume of water (Q) that will move in a unit

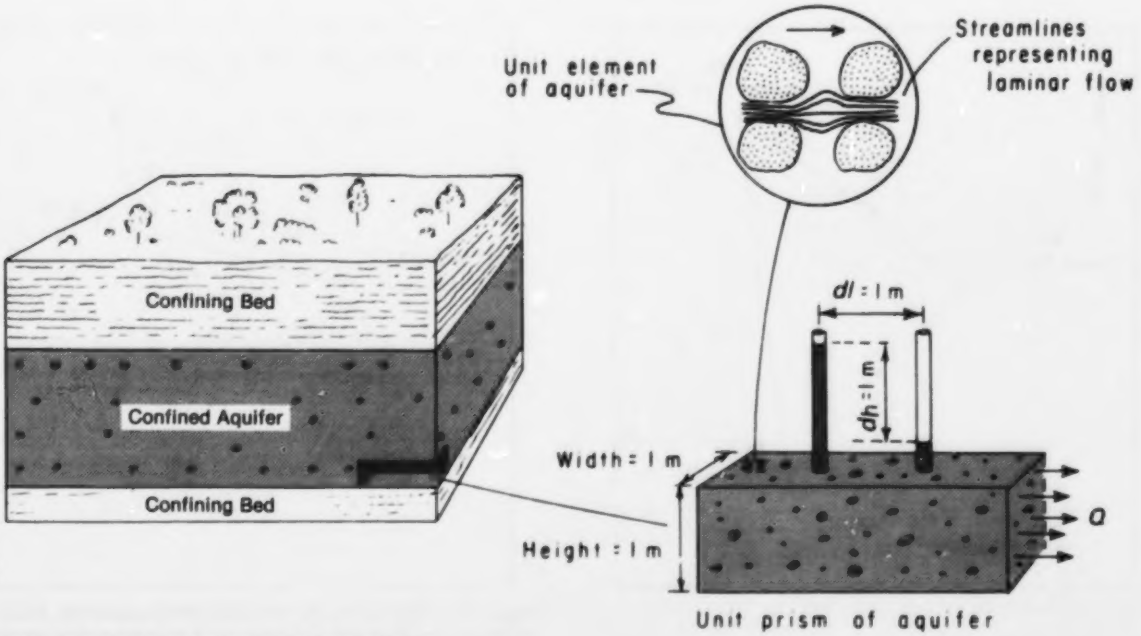


Figure 12. Factors Affecting Hydraulic Conductivity

of time (commonly, a day) under a unit hydraulic gradient (such as a meter per meter) through a unit area (such as a square meter). Figure 12 illustrates these factors. Expressing hydraulic conductivity in terms of a unit gradient, rather than of an actual gradient at some place in an aquifer, permits ready comparison of values of hydraulic conductivity for different rocks.

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[1] Where hydraulic gradient is discussed as an independent entity, as it is in "Heads and Gradients," it is shown symbolically as h_L/L and is referred to as head loss per unit of distance. Where hydraulic gradient appears as one of the factors in an equation, as it does in equation 6, it is shown symbolically as dh/dl to be consistent with other ground-water literature. The gradient dh/dl indicates that the unit distance is reduced to as small a value as one can imagine, in accordance with the concepts of differential calculus.

Hydraulic conductivity replaces the term "field coefficient of permeability" and should be used in referring to the water-transmitting characteristic of material in quantitative terms. It is still common practice to refer in qualitative terms to "permeable" and "impermeable" material.

Figure 13 shows that the hydraulic conductivity of rocks ranges through 12 orders of magnitude. There are few physical parameters whose values range so widely. Hydraulic conductivity is not only different in different types of rocks but may also be different from place to place in the same rock. If the hydraulic conductivity is essentially the same in any area, the aquifer in that area is said to be *homogeneous*. If, on the other hand, the hydraulic conductivity differs from one part of the area to another, the aquifer is said to be *heterogeneous*.

Hydraulic conductivity may also be different in different directions at any place in an aquifer. If the hydraulic conductivity is essentially the same in all

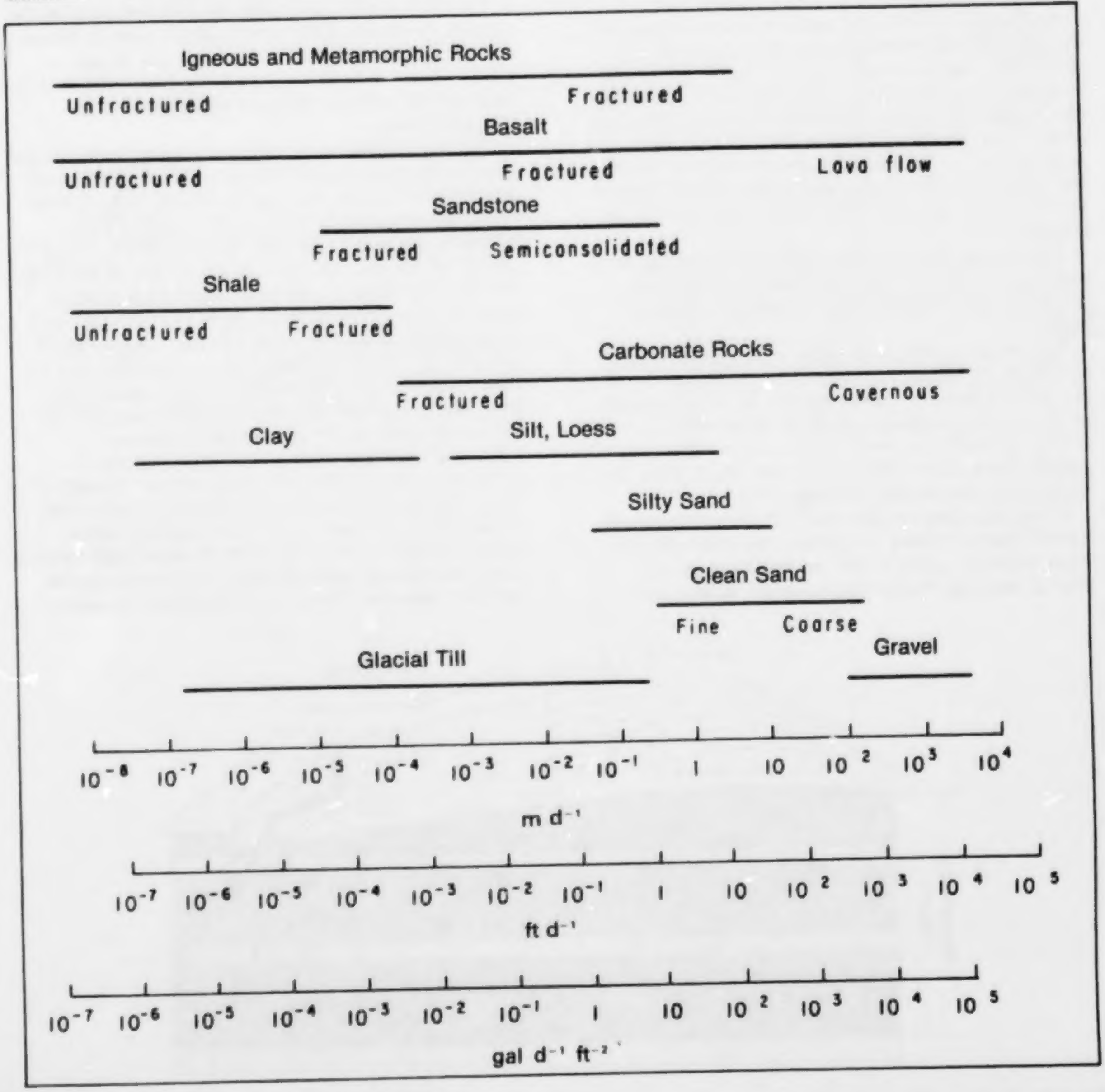


Figure 13. Hydraulic Conductivity of Selected Rocks

directions, the aquifer is said to be *isotropic*. If it is different in different directions, the aquifer is said to be *anisotropic*.

Although it is convenient in many mathematical analyses of ground-water flow to assume that aquifers are both homogeneous and isotropic, such aquifers are rare, if they exist at all. The condition most commonly encountered is for hydraulic conductivity in most rocks and especially in unconsolidated deposits and in flat-lying consolidated sedimentary rocks to be larger in the horizontal direction than it is in the vertical direction.

Functions of Ground-Water Systems

As Figure 14 shows, the aquifers and confining beds that underlie any area comprise the *ground-water system* of the area. Hydraulically, this system serves two functions: it stores water to the extent of its porosity, and it transmits water from recharge areas to discharge areas. Thus, a ground-water system serves as both a reservoir and a conduit. With the exception of cavernous limestones, lava flows, and coarse gravels, ground-water systems are more effective as reservoirs than as conduits.

Water enters ground-water systems in *recharge areas* and moves through them, as dictated by hydraulic gradients and hydraulic conductivities, to *discharge areas*.

The identification of recharge areas is becoming increasingly important because of the expanding use of the land surface for waste disposal. In the humid part of the country, recharge occurs in all interstream areas —that is, in all areas except along streams and their adjoining flood plains. The streams and flood plains are, under most conditions, discharge areas.

In the drier part (western half) of the conterminous United States, recharge conditions are more complex. Most recharge occurs in the mountain ranges, on alluvial fans that border the mountain ranges, and along the channels of major streams where they are underlain by thick and permeable alluvial deposits.

Recharge rates are generally expressed in terms of volume (such as cubic meters or gallons) per unit of time (such as a day or a year) per unit of area (such as a square kilometer, a square mile, or an acre). When these units are reduced to their simplest forms, the result is recharge expressed as a depth of water on the land surface per unit of time. Recharge varies from year to year, depending on the amount of precipitation, its seasonal distribution, air temperature, land use, and other factors. Relative to land use, recharge rates in forests are much higher than those in cities.

Annual recharge rates range, in different parts of the country, from essentially zero in desert areas to about 600 mm yr^{-1} (1,600 m^{3}km^{-2}d^{-1} or 1.1×10^6 gal mi^{-2}d^{-1}) in the rural areas on Long Island and in other rural areas in the East that are underlain by very permeable soils.

The rate of movement of ground water from recharge areas to discharge areas depends on the hydraulic conductivities of the aquifers and confining beds, if water moves downward into other aquifers, and on the hydraulic gradients. (See the "Ground-Water Velocity" section of this chapter.) A convenient way of showing the rate is in terms of the time required for ground-water to move from different parts of a recharge area to the nearest discharge area. The time ranges from a few days in the zone adjacent to the discharge area to thousands of years (millennia) for water that moves from the central part of some recharge areas through the deeper parts of the ground-water system.

Natural discharge from ground-water systems includes not only the flow of springs and the seepage of water into stream channels or wetlands but also evaporation from the upper part of the capillary fringe, where it occurs within a meter or so of the land surface. Large amounts of water are also withdrawn from the capillary fringe and the zone of saturation by plants

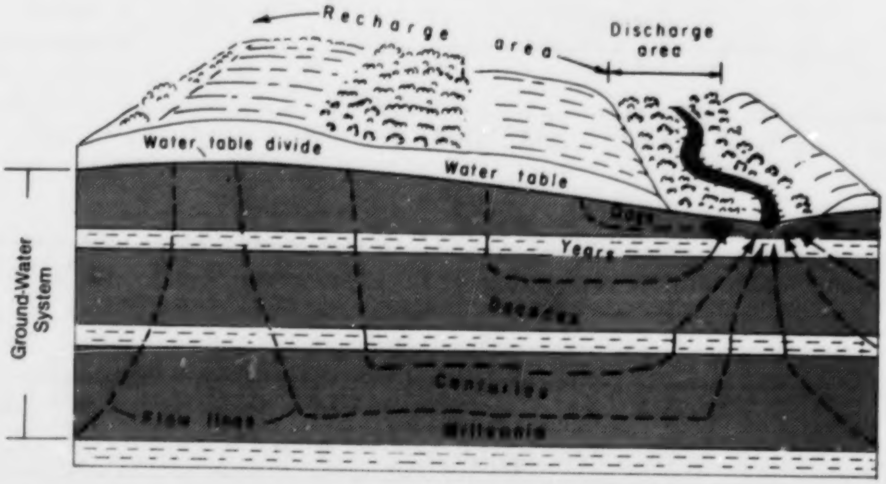


Figure 14. The Ground-Water System

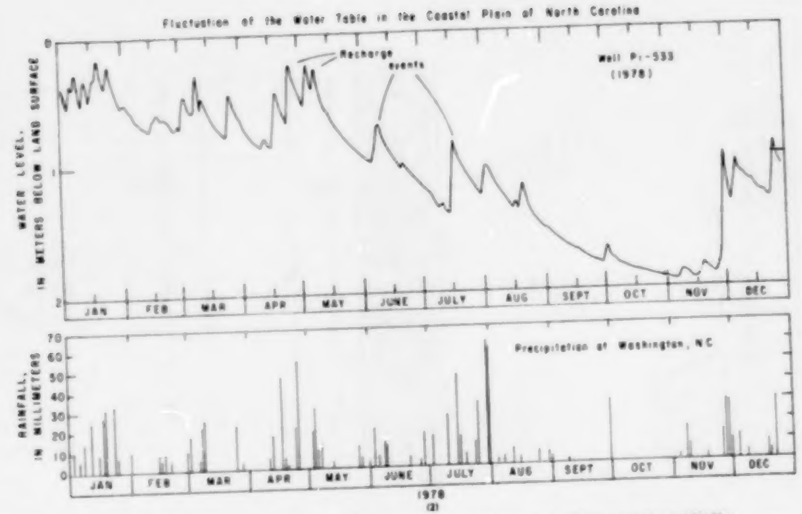


Figure 15. Fluctuation of the Water Level Showing Recharge and Discharge in Observation Wells.

during the growing season. Thus, discharge areas include not only the channels of perennial streams but also the adjoining flood plains and other low-lying areas.

One of the most significant differences between recharge areas and discharge areas is that the areal extent of discharge areas is invariably much smaller than that of recharge areas. This size difference shows, as we would expect, that discharge areas are more "efficient" than recharge areas. Recharge involves unsaturated movement of water in the vertical direction; in other words, movement is in the direction in which the hydraulic conductivity is generally the lowest. Discharge, on the other hand, involves saturated movement, much of it in the horizontal direction—that is, in the direction of the largest hydraulic conductivity.

Another important aspect of recharge and discharge involves timing. Recharge occurs during and immediately following periods of precipitation and thus is intermittent. Discharge, on the other hand, is a continuous process as long as ground-water heads are above the level at which discharge occurs. However, between periods of recharge, ground-water heads decline, and the rate of discharge also declines. Most recharge of ground-water systems occurs during late fall, winter, and early spring, when plants are dormant and evaporation rates are small. These aspects of recharge and discharge are apparent from graphs showing the fluctuation of the water level in observation wells, such as the one shown in Figure 15. The occasional lack of correlation, especially in the summer, between the precipitation and the rise in water level is due partly to the distance of 20 km between the weather station and the well.

Capillarity and Unsaturated Flow

Most recharge of ground-water systems occurs during the percolation of water across the unsaturated zone. The movement of water in the unsaturated zone is controlled by both gravitational and capillary forces.

Capillarity results from two forces: the mutual attraction (cohesion) between water molecules and the molecular attraction (adhesion) between water and different solid materials. Figure 16 shows that a consequence of these forces, water will rise in small-diameter glass tubes to a height h_c above the water level in a large container.

Most pores in granular materials are of capillary size, and as a result, water is pulled upward into a capillary fringe above the water table in the same manner that water would be pulled up into a column of sand whose lower end is immersed in water, as Figure 17 shows. Table 4 shows the approximate capillary rise in selected granular materials.

Table 4. Approximate Height of Capillary Rise (h_c) in Granular Materials

Material	Rise (mm)
Sand:	125
Coarse	250
Medium	400
Fine	1,000
Silt	

Steady-state flow of water in the unsaturated zone can be determined from a modified form of Darcy's law. Steady state in this context refers to a condition in which the moisture content remains constant, as it would, for example, beneath a waste-disposal pond whose bottom is separated from the water table by an unsaturated zone.

Steady-state unsaturated flow (Q) is proportional to the effective hydraulic conductivity (K_e), the cross-sectional area (A) through which the flow occurs, and the gradients due to both capillary forces and gravitational forces. Thus,

$$Q = K_e A \left(\frac{h_c - z}{z}\right) \pm \left(\frac{dh}{dl}\right) \qquad (8)$$

where Q is the quantity of water, K_e is the hydraulic conductivity under the degree of saturation existing in

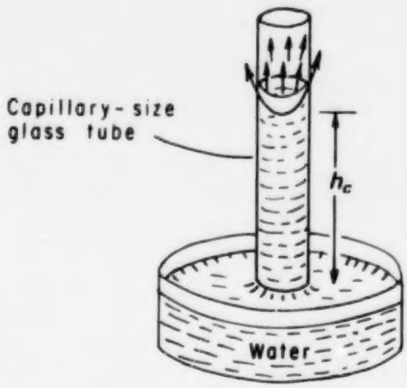


Figure 16. Capillarity

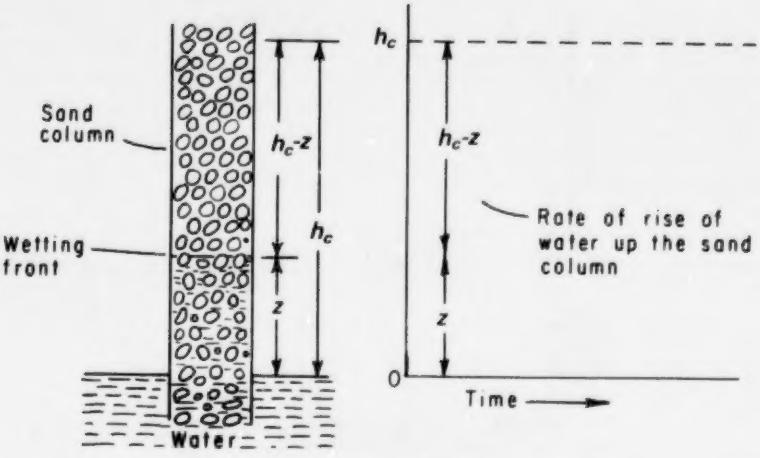


Figure 17. Capillary Action in a Column of Sand

the unsaturated zone, $(h_c - z)/z$ is the gradient due to capillary (surface tension) forces, and dh/dl is the gradient due to gravity.

The plus or minus sign is related to the direction of movement—plus for downward and minus for upward. For movement in a vertical direction, either up or down, the gradient due to gravity is 1/1, or 1. For lateral (horizontal) movement in the unsaturated zone, the term for the gravitational gradient can be eliminated.

Figure 17 shows that the capillary gradient at any time depends on the length of the water column (z) supported by capillarity in relation to the maximum possible height of capillary rise (h_c). For example, if the lower end of a sand column is suddenly submerged in water, the capillary gradient is at a maximum, and the rate of rise of water is fastest. As the wetting front advances up the column, the capillary gradient declines, and the rate of rise decreases.

The capillary gradient can be determined from tensiometer measurements of hydraulic pressures. To determine the gradient, Figure 18 shows it is necessary to measure the negative pressures (h_p) at two levels in the unsaturated zone. The equation for total head (h_t) is

$$h_t = z + h_p \tag{5}$$

where z is the elevation of a tensiometer. Substituting values in this equation for tensiometer No. 1 in Figure 18, we obtain

$$h_t = 32 + (-1) = 32 - 1\,\text{d} = 31\ \text{m}$$

The total head at tensiometer No. 2 if 26 m. The vertical distance between the tensiometers is 32 m minus 28 m, or 4 m. Because the combined gravitational and capillary hydraulic gradient equals the head loss divided by the distance between tensiometers, the gradient is

$$\frac{h_L}{L} = \frac{h_{t(1)} - h_{t(2)}}{z_{(1)} - z_{(2)}} = \frac{31 - 26}{32 - 28} = \frac{5\ \text{m}}{4\ \text{m}} = 1.25$$

This gradient includes both the gravitational gradient (dh/dl) and the capillary gradient ($[h_c - z]/z$). Because the head in tensiometer No. 1 exceeds that in tensiometer No. 2, we know that flow is vertically downward and that the gravitational gradient is 1/1, or 1. Therefore, the capillary gradient is 0.25 m m^{-1} (1.25 – 1.00).

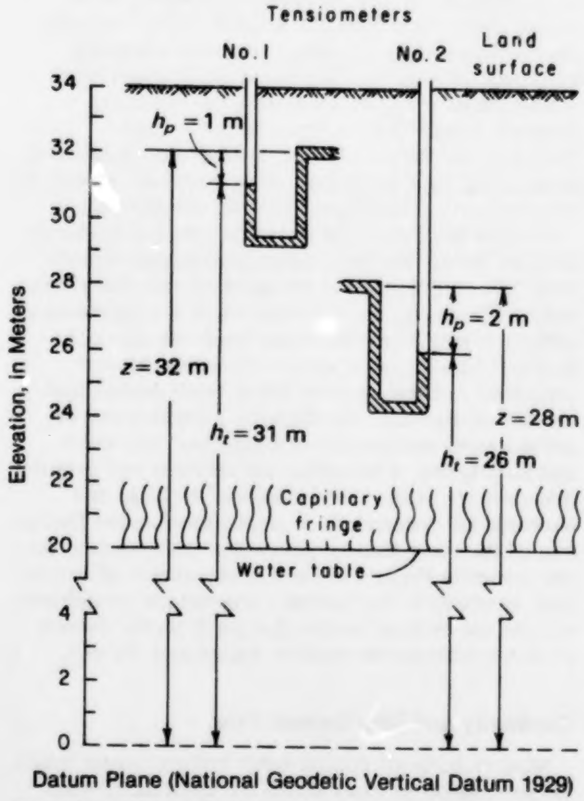


Figure 18. Tensiometer Measurements for Measuring the Capillary Gradient

The effective hydraulic conductivity (K_e) is the hydraulic conductivity of material that is not completely saturated. It is thus less than the (saturated) hydraulic conductivity (K_s) for the material. Figure 19 shows the relation between degree of saturation and the ratio of saturated and unsaturated hydraulic conductivity for coarse sand. The hydraulic conductivity (K_s) of coarse sand is about 60 m d^{-1}.

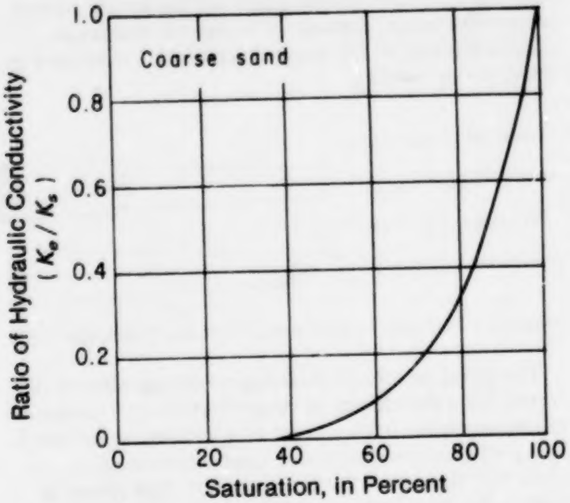


Figure 19. Relation Between Degree of Saturation and the Ratio of Saturated and Unsaturated Hydraulic Conductivity for Coarse Sand

Stratification and Unsaturated Flow

Most sediments are deposited in layers (beds) that have a distinct grain size, sorting, or mineral composition. Where adjacent layers differ in one of these characteristics or more, the deposit is said to be *stratified,* and its layered structure is referred to as *stratification.*

The layers comprising a stratified deposit commonly differ from one another in both grain size and sorting and, consequently, differ from one another in hydraulic conductivity. These differences in hydraulic conductivity significantly affect both the percolation of water across the unsaturated zone and the movement of groundwater.

In most areas, the unsaturated zone is composed of horizontal or nearly horizontal layers. The movement of water, on the other hand, is predominantly in a vertical direction. In many ground-water problems, and especially in those related to the release of pollutants at the land surface, the effect of stratification on movement of fluids across the unsaturated zone is of great importance.

The manner in which water moves across the unsaturated zone has been studied by using models containing glass beads. Figure 20 illustrates one model containing beads of a single size representing a nonstratified deposit, and Figure 21 shows another model consisting of five layers, three of which were finer grained and more impermeable than the other two. The dimensions of the models were about 1.5 m × 1.2 m × 76 mm.

In the nonstratified model, water introduced at the top moved vertically downward through a zone of con-

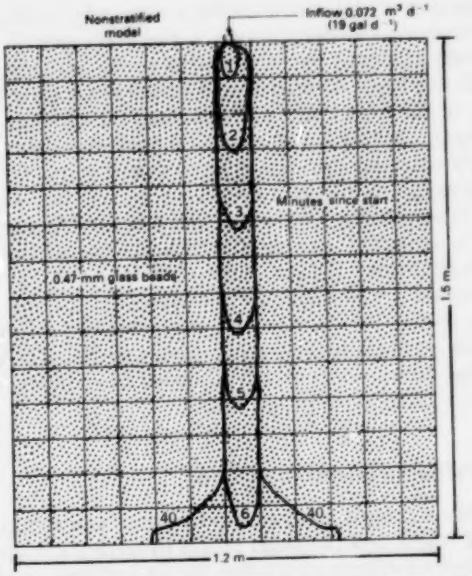


Figure 20. Single-Size Bead Model Illustrating Water Movement Across the Unsaturated Zone.

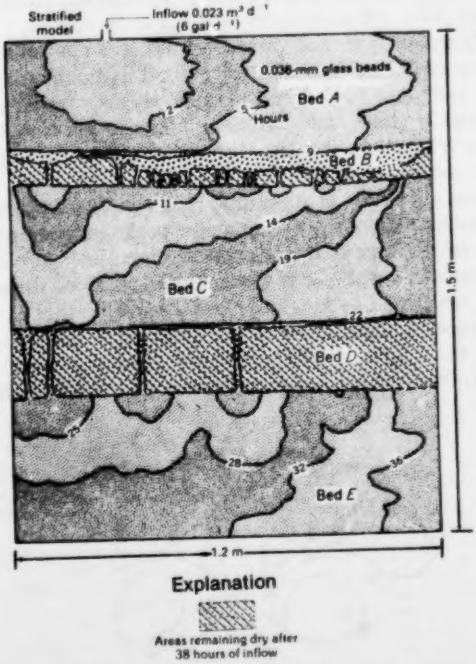


Figure 21. Five-Layer Model Illustrating Water Movement Across the Unsaturated Zone.

stant width to the bottom of the model. In the stratified model, beds *A*, *C*, and *E* consisted of silt-sized beads (diameters of 0.036 mm) having a capillary height (h_c) of about 1,000 mm and a hydraulic conductivity (K) of 0.8 m d^{-1}. Beds *B* and *D* consisted of medium-sand-sized beads (diameters of 0.47 mm) having a capillary height of about 250 mm and a hydraulic conductivity of 82 m d^{-1}.

Because of the strong capillary force and the low hydraulic conductivity in bed *A*, the water spread laterally at almost the same rate as it did vertically, and it did not begin to enter bed *B* until 9 hours after the start of the experiment. At that time, the capillary saturation in bed *A* had reached a level where the unsatisfied (remaining) capillary pull in bed *A* was the same as that in bed *B*. In other words, z in bed *A* at that time equaled 1,000 mm – 250 mm, or 750 mm. (For a definition of z, see the "Capillarity and Unsaturated Flow" section of this Chapter.)

Because the hydraulic conductivity of bed *B* was 100 times that of bed *A*, water moved across bed *B* through narrow vertical zones. We can guess that the glass beads in these zones were packed somewhat more tightly than those in other parts of the beds.

Saturated Flow and Dispersion

In the saturated zone, all interconnected openings are full of water, and the water moves through these openings in the direction controlled by the hydraulic gradient. Movement in the saturated zone may be either laminar or turbulent. In *laminar flow,* water particles move in an orderly manner along streamlines. In *turbulent flow,* water particles move in a disordered, highly irregular manner, which results in a complex mixing of the particles. Under natural hydraulic gradients, turbulent flow occurs only in large openings such as those in gravel, lava flows, and limestone caverns. Flows are laminar in most granular deposits and fractured rocks.

In laminar flow in a granular medium, the different streamlines converge in the narrow necks between particles and diverge in the larger interstices, as Figure 22 illustrates. Thus, there is some intermingling of streamlines, which results in transverse dispersion—that is, dispersion at right angles to the direction of ground-water flow. Also, differences in velocity result from friction between the water and the rock particles. The slowest rate of movement occurs adjacent to the particles, and the fastest rate occurs in the center of pores. The resulting dispersion is longitudinal—that is, in the direction of flow.

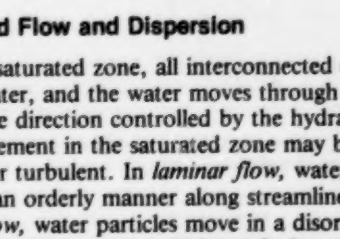


Figure 22. Dispersion in a Granular Deposit

As Figure 23 shows, Danel (1953) found that dye injected at a point in a homogeneous and isotropic granular medium dispersed laterally in the shape of a cone about 6° wide. He also found that the concentration of dye over a plane at any given distance from the inlet point is a bell-shaped curve similar to the normal probability curve. Because of transverse and longitudinal dispersion, the peak concentration decreased in the direction of flow.

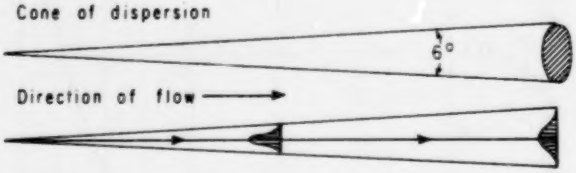


Figure 23. Changes in Concentration in the Dispersion Cone

The effect of longitudinal dispersion can also be observed from the change in concentration of a substance (C) downstream from a point at which the substance is being injected constantly at a concentration of C_o. As Figure 24 illustrates, the concentration rises slowly at first as the "fastest" streamlines arrive and then rises rapidly until the concentration reaches about 0.7 C_o, at which point the rate of increase in concentration begins to decrease.

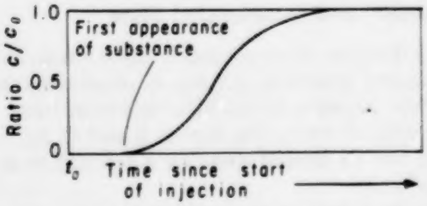


Figure 24. Effect of Longitudinal Dispersion

Dispersion is important in the study of ground-water pollution. However, it is difficult to measure in the field because the rate and direction of movement of wastes are also affected by stratification, ion exchange, filtration, and other conditions and processes. Stratification and areal differences in lithology and other characteristics of aquifers and confining beds actually result in much greater lateral and longitudinal dispersion than that measured by Danel for a homogeneous and isotropic medium.

Ground-Water Movement and Topography

It is desirable, wherever possible, to determine the position of the water table and the direction of ground-water movement. To do so, it is necessary to determine the altitude, or the height above a datum plane, of the water level in wells. However, in most areas, general but very valuable conclusions about the direction of ground-water movement can be derived from observations of land-surface topography, as Figure 25 illustrates.

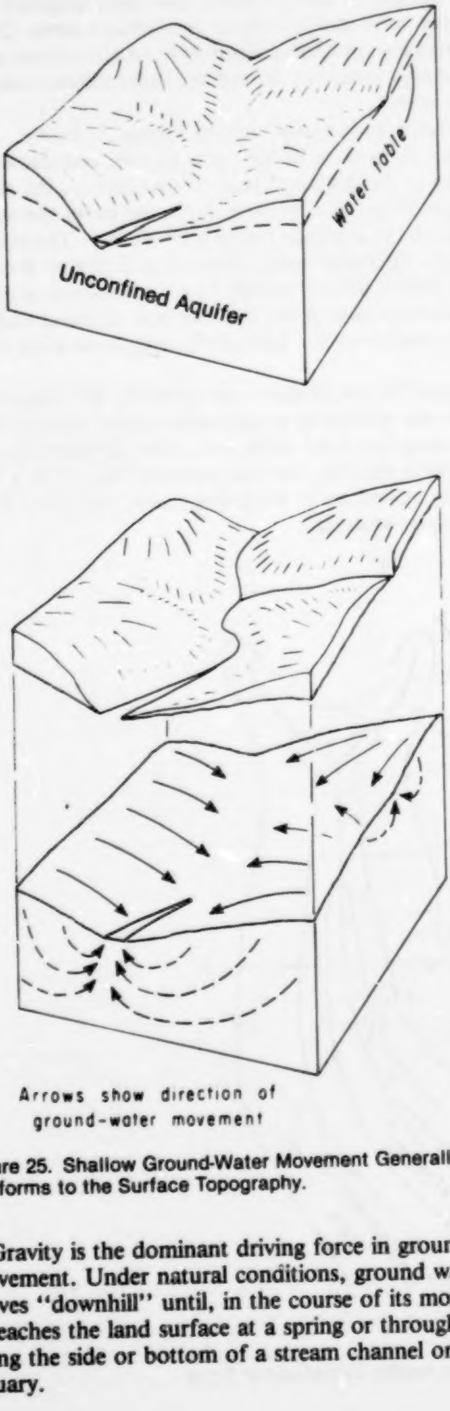


Figure 25. Shallow Ground-Water Movement Generally Conforms to the Surface Topography.

Gravity is the dominant driving force in ground-water movement. Under natural conditions, ground water moves "downhill" until, in the course of its movement, it reaches the land surface at a spring or through a seep along the side or bottom of a stream channel or an estuary.

Thus, ground water in the shallowest part of the saturated zone moves from interstream areas toward streams or the coast. If we ignore minor surface irregularities, we find that the slope of the land surface is also toward streams or the coast. The depth to the water table is greater along the divide between streams than it is beneath the flood plain. In effect, the water table usually is a subdued replica of the land surface.

In areas where ground water is used for domestic and other needs requiring good-quality water, septic tanks, sanitary landfills, waste ponds, and other waste-disposal sites should not be located uphill from supply wells.

The potentiometric surface of confined aquifers, like the water table, also slopes from recharge areas to discharge areas. Shallow confined aquifers, which are relatively common along the Atlantic Coastal Plain, share both recharge and discharge areas with the surficial unconfined aquifers. This sharing may not be the case with the deeper confined aquifers. The principal recharge areas for these are probably in their outcrop areas near the western border of the Coastal Plain, and their discharge areas are probably near the heads of the estuaries along the major streams. Thus, movement of water through these aquifers is in a general west to east direction, where it has not been modified by withdrawals.

In the western part of the conterminous United States, and especially in the alluvial basins region, conditions are more variable than those described above. In this area, streams flowing from mountain ranges onto alluvial plains lose water to the alluvial deposits; thus, ground water in the upper part of the saturated zone flows down the valleys and at an angle away from the streams.

Ground water is normally hidden from view; as a consequence, many people have difficulty visualizing its occurrence and movement. This difficulty adversely affects their ability to understand and to deal effectively with ground-water-related problems. This problem can be partly solved through the use of flow nets, which are one of the most effective means yet devised for illustrating conditions in ground-water systems.

Ground-Water Flow Nets

Flow nets consist of two sets of lines. One set, referred to as *equipotential lines,* connects points of equal head and thus represents the height of the water table, or the potentiometric surface of a confined aquifer, above a datum plane. The second set, referred to as *flow lines,* depicts the idealized paths followed by particles of water as they move through the aquifer. Because ground water moves in the direction of the steepest hydraulic gradient, flow lines in isotropic aquifers are perpendicular to equipotential lines—that is, flow lines cross equipotential lines at right angles.

There are an infinite number of equipotential lines and flow lines in an aquifer. However, for purposes of flow-net analysis, only a few of each set need be drawn. Equipotential lines are drawn so that the drop in head is the same between adjacent pairs of lines. Flow lines are drawn so that the flow is equally divided between adjacent pairs of lines and so that, together with the equipotential lines, they form a series of "squares."

Flow nets not only show the direction of ground-water movement but can also, if they are drawn with care, be used to estimate the quantity of water in transit through an aquifer. According to Darcy's law, the flow through any "square" is

$$q = Kbw\left(\frac{dh}{dl}\right) \qquad (9)$$

and the total flow through any set or group of "squares" is

$$Q = nq \qquad (10)$$

where K is hydraulic conductivity, b is aquifer thickness at the midpoint between equipotential lines, w is the distance between flow lines, dh is the difference in head between equipotential lines, dl is the distance between equipotential lines, and n is the number of squares through which the flow occurs.

Figures 26 and 27 show a flow net in both plan view and cross section for an area underlain by an unconfined aquifer composed of sand. The sand overlies a horizontal confining bed, the top of which occurs at an elevation 3 m above the datum plane. The fact that some flow lines originate in the area in which heads exceed 13 m indicates the presence of recharge to the aquifer in this area. The relative positions of the land surface and the water table in Figure 27 suggest that recharge occurs throughout the area, except along the stream valleys. This suggestion is confirmed by the fact that flow lines also originate in areas where heads are less than 13 m.

As Figures 26 and 27 show, flow lines originate in recharge areas and terminate in discharge areas. Closed contours (equipotential lines) indicate the central parts of recharge areas but do not normally indicate the limits of the areas.

In the cross-sectional view in Figure 27, heads decrease downward in the recharge area and decrease upward in the discharge area. Consequently, the deeper a well is drilled in a recharge area, the lower the water level in the well stands below land surface. The reverse is true in discharge areas. Thus, in a discharge area, if a well is drilled deeply enough in an unconfined aquifer, the well may flow above land surface. Consequently, a flowing well does not necessarily indicate artesian conditions.

Figures 28 and 29 show equipotential lines and flow lines in the vicinity of a stream that gains water in its headwaters and loses water as it flows downstream. In the gaining reaches, the equipotential lines form a V pointing upstream; in the losing reach, they form a V pointing downstream.

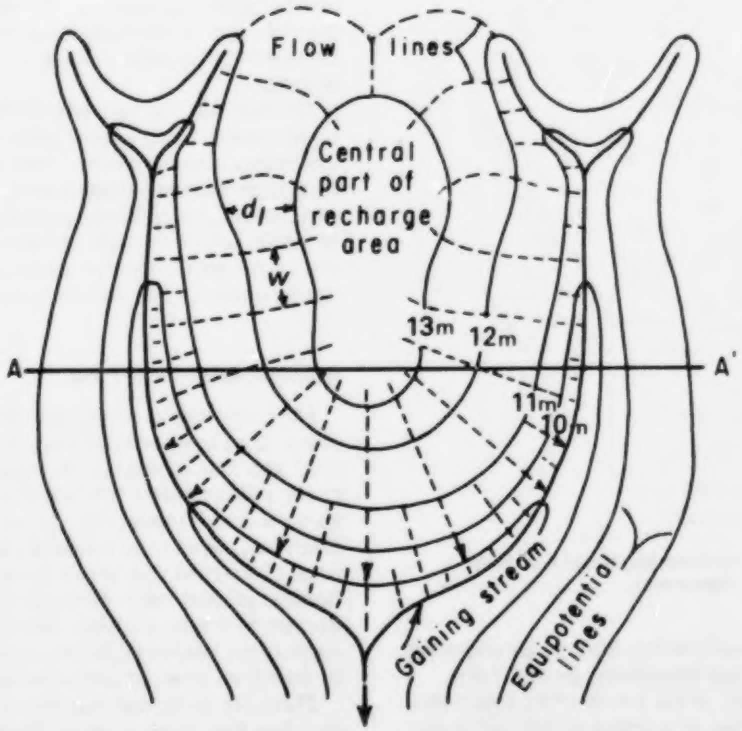


Figure 26. Plan View of the Flow Net for an Area Underlain by an Unconfined Aquifer Composed of Sand.

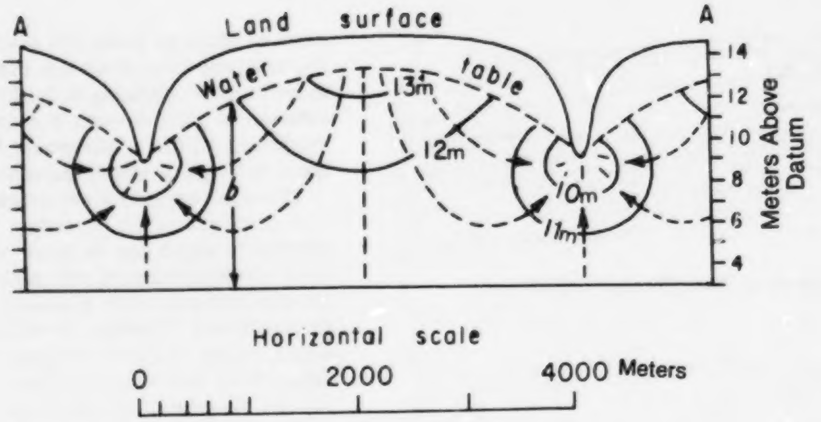


Figure 27. Cross Section of the Flow Net in Figure 26.

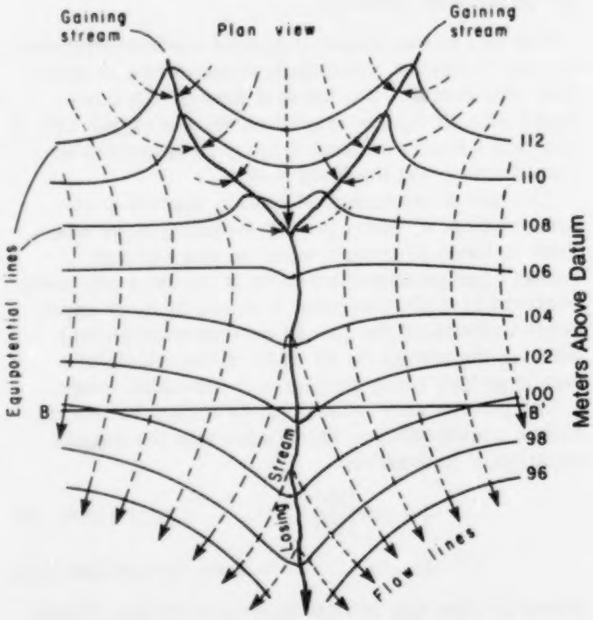


Figure 28. Plan View of the Flow Net for the Vicinity of a Stream that Gains Water in its Headwaters and Loses Water as it Flows Downstream.

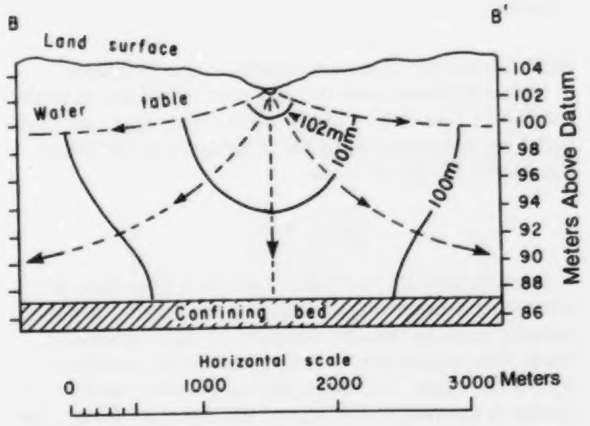


Figure 29. Cross Section of the Flow Net in Figure 28.

Ground-Water Movement and Stratification

Nearly all ground-water systems include both aquifers and confining beds. Figure 30 shows that ground-water movement through these systems involves flow not only *through* the aquifers but also *across* the confining beds.

The hydraulic conductivities of aquifers are tens to thousands of times those of confining beds. Thus, aquifers offer the least resistance to flow, the result being that, for a given rate of flow, the head loss per unit of distance along a flow line is tens to thousands of times less in aquifers than it is in confining beds. Consequently, as Figure 31 shows, lateral flow in confining beds usually is negligible, and flow lines tend to "concentrate" in aquifers and be parallel to aquifer boundaries.

Differences in the hydraulic conductivities of aquifers and confining beds cause a refraction or bending of flow lines at their boundaries. As flow lines move from aquifers into confining beds, they are refracted toward the direction perpendicular to the boundary. In other words, they are refracted in the direction that produces the shortest flow path in the confining bed. As the flow lines emerge from the confining bed, they are refracted

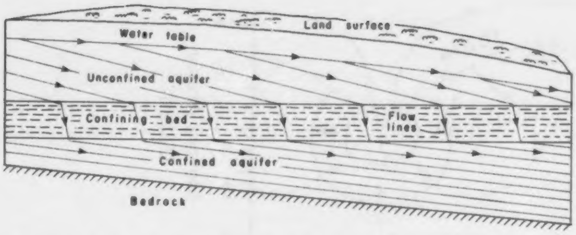


Figure 30. Movement of Water Through Ground-Water Systems

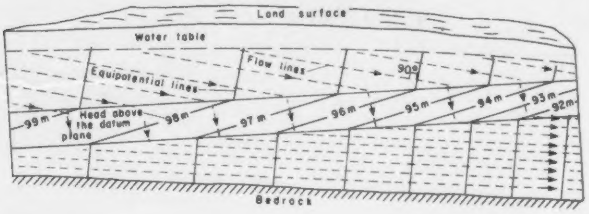


Figure 31. Concentration of Flow Lines in Ground-Water Systems

back toward the direction parallel to the boundary.

Figure 32 shows that the angles of refraction (and the spacing of flow lines in adjacent aquifers and confining beds) are proportional to the differences in hydraulic conductivities (K) such that

$$\frac{\tan \theta_1}{\tan \theta_2} = \frac{K_1}{K_2} \qquad (11)$$

In cross section, the water table is a flow line. It represents a bounding surface for the ground-water system; thus, in the development of many ground-water flow equations, it is assumed to be coincident with a flow line. However, during periods when recharge is arriving at the top of the capillary fringe, the water table is also the point of origin of flow lines, as Figure 30 shows.

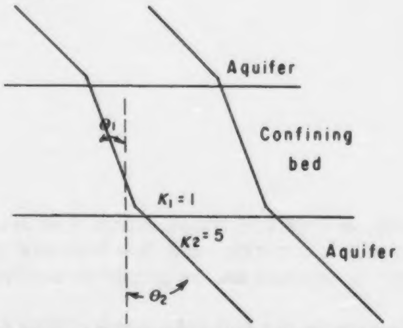


Figure 32. Angles of Refraction in Ground-Water Systems

The movement of water through ground-water systems is controlled by the vertical and horizontal hydraulic conductivities and thicknesses of the aquifers and confining beds and the hydraulic gradients. The maximum difference in head exists between the central parts of recharge areas and discharge areas. Because of the relatively large head loss that occurs as water moves across confining beds, the most vigorous circulation of ground water normally occurs through the shallowest aquifers. Movement becomes more and more lethargic as depth increases.

The most important exceptions to the general situation described in the proceding paragraph are those systems in which one or more of the deeper aquifers have transmissivities significantly larger than those of the surficial and other shallower aquifers. Thus, in eastern North Carolina, the Castle Hayne Limestone, which occurs at depths ranging from about 10 to about 75 m below land surface, is the dominant aquifer because of its very large transmissivity, although it is overlain in most of the area by one or more less permeable aquifers.

Ground-Water Velocity

The rate of movement of ground water is important in many problems, particularly those related to pollution. For example, if a harmful substance is introduced into an aquifer upgradient from a supply well, it becomes a matter of great urgency to estimate when the substance will reach the well.

The rate of movement of ground water is greatly overestimated by many people, including those who think in terms of ground water moving through "veins" and underground rivers at the rates commonly observed in surface streams. It would by more appropriate to compare the rate of movement of ground water to the movement of water in the middle of a very large lake being drained by a very small stream.

The *ground-water velocity* equation can be derived from a combination of Darcy's law and the velocity equation of hydraulics.

$$Q = KA\left(\frac{dh}{dl}\right) \qquad \text{(Darcy's law)} \quad (6)$$

$$Q = Av \qquad \text{(velocity equation)} \quad (12)$$

where Q is the rate of flow or volume per unit of time, K is the hydraulic conductivity, A is the cross-sectional area, at a right angle to the flow direction, through which the flow Q occurs, dh/dl is the hydraulic gradient, and v is the Darcian velocity, which is the average velocity of the entire cross-sectional area. Combining equations 6 and 12, we obtain

$$Av = KA\left(\frac{dh}{dl}\right) \qquad (13)$$

Canceling the area terms, we find that

$$v = K\left(\frac{dh}{dl}\right)$$

Because this equation contains terms for hydraulic conductivity and gradient only, it is not yet a complete expression of ground-water velocity. The missing term is porosity (n) because, as we know, water moves only through the openings in a rock. Adding the porosity term, we obtain

$$v = \frac{Kdh}{ndl} \tag{14}$$

In order to demonstrate the relatively slow rate of ground-water movement, equation 14 is used to determine the rate of movement through an aquifer and a confining bed.

1. *Aquifer composed of coarse sand*

$K = 60$ m/d

$dh/dl = 1$ m/1,000 m

$n = 0.20$

$$v = \frac{K}{n} \times \frac{dh}{dl} = \frac{60 \text{ m}}{d} \times \frac{1}{0.20} \times \frac{1 \text{ m}}{1{,}000 \text{ m}}$$

$$= \frac{60 \text{ m}^2}{200 \text{ m d}} = 0.3 \text{ m d}^{-1}$$

2. *Confining bed composed of clay*

$K = 0.0001$ m/d

$dh/dl = 1$ m/10 m

$n = 0.50$

$$v = \frac{0.0001 \text{ m}}{\text{d}} \times \frac{1}{0.50} \times \frac{1 \text{ m}}{10 \text{ m}}$$

$$= \frac{0.0001 \text{ m}^2}{5 \text{ m d}} = 0.00002 \text{ m d}^{-1}$$

Velocities calculated with equation 14 are, at best, average values. Where ground-water pollution is involved, the fastest rates of movement may be several times the average rate. Also, the rates of movement in limestone caverns, lava tubes, and large rock fractures may approach those observed in surface streams.

Further, movement in unconfined aquifers is not limited to the zone below the water table or to the saturated zone. Water in the capillary fringe is subjected to the same hydraulic gradient that exists at the water table; water in the capillary fringe moves, therefore, in the same direction as the ground water.

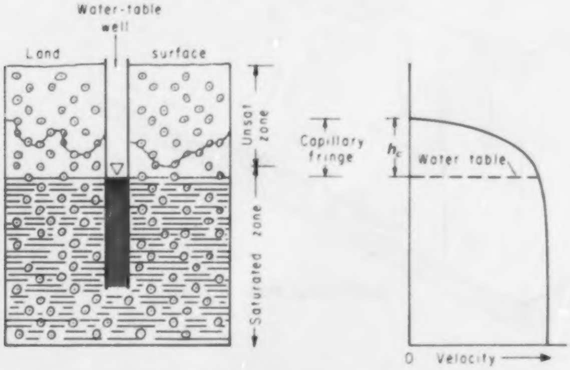


Figure 33. Rate of Lateral Movement in the Capillary Fringe

As Figure 33 shows, the rate of lateral movement in the capillary fringe decreases in an upward direction and becomes zero at the top of the fringe. This consideration is important where unconfined aquifers are polluted with gasoline and other substances less dense than water.

Transmissivity

The capacity of an aquifer to transmit water of the prevailing kinematic viscosity is referred to as its transmissivity. The *transmissivity* (T) of an aquifer is equal to the hydraulic conductivity of the aquifer multipled by the saturated thickness of the aquifer. Thus,

$$T = Kb \tag{15}$$

where T is transmissivity, K is hydraulic conductivity, and b is aquifer thickness.

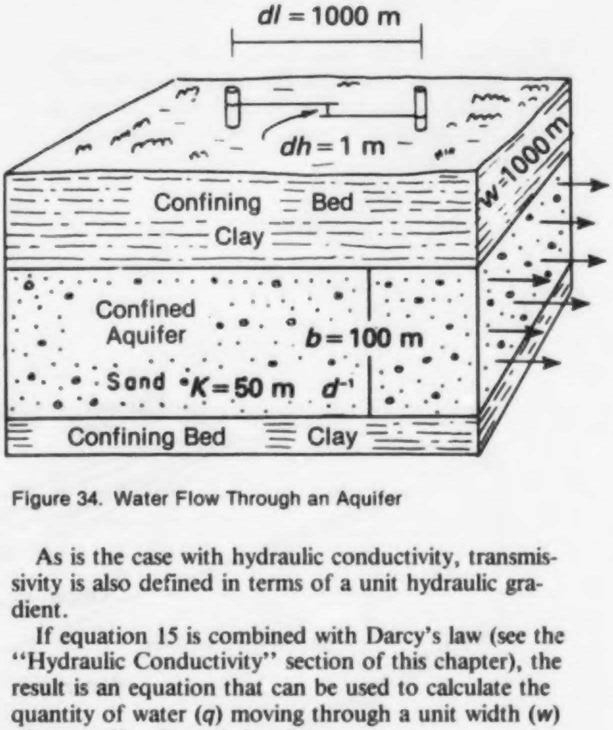


Figure 34. Water Flow Through an Aquifer

As is the case with hydraulic conductivity, transmissivity is also defined in terms of a unit hydraulic gradient.

If equation 15 is combined with Darcy's law (see the "Hydraulic Conductivity" section of this chapter), the result is an equation that can be used to calculate the quantity of water (q) moving through a unit width (w) of an aquifer. Darcy's law is

$$q = KA\left(\frac{dh}{dl}\right) \tag{6a}$$

Expressing area (A) as bw, we obtain

$$q = Kbw\left(\frac{dh}{dl}\right) \tag{9}$$

Next, expressing transmissivity (T) as Kb, we obtain

$$q = Tw\left(\frac{dh}{dl}\right) \tag{16}$$

Equation 16 modified to determine the quantity of water (Q) moving through a large width (W) of an aquifer is

$$Q = TwW\left(\frac{dh}{dl}\right) \tag{17}$$

or, if it is recognized that T applies to a unit width (w) of an aquifer, this equation can be stated more simply as

$$Q = TW\left(\frac{dh}{dl}\right) \tag{18}$$

If equation 18 is applied to Figure 34, the quantity of water flowing out of the right-hand side of the figure can be calculated by using the values shown on the figure as follows:

$$T = Kb = \frac{50\ \mathrm{m}}{\mathrm{d}} \times \frac{100\ \mathrm{m}}{1} = 5{,}000\ \mathrm{m^2 d^{-1}}$$

$$Q = TW\left(\frac{dh}{dl}\right) = \frac{5{,}000\ \mathrm{m^2}}{\mathrm{d}} \times \frac{1{,}000\ \mathrm{m}}{1} \times \frac{1\mathrm{m}}{1{,}000\ \mathrm{m}} = 5{,}000\ \mathrm{m^3\ d^{-1}}$$

Equation 18 is also used to calculate transmissivity, where the quantity of water (Q) discharging from a known width of aquifer can be determined as, for example, with streamflow measurements. Rearranging terms, we obtain

$$T = \frac{Q}{W}\left(\frac{dl}{dh}\right) \tag{19}$$

The units of transmissivity, as the preceding equation demonstrates, are

$$T = \frac{(\mathrm{m^3\ d^{-1}})(\mathrm{m})}{(\mathrm{m})(\mathrm{m})} = \frac{\mathrm{m^2}}{\mathrm{d}}$$

Figure 35 illustrates the hydrologic situation that permits calculation of transmissivity through the use of stream discharge. The calculation can be made only during dry-weather (baseflow) periods, when all water in the stream is derived from ground-water discharge. For the purpose of this example, the following values are assumed:

Average daily flow at stream-gaging station A:	2.485 $\mathrm{m^3\ s^{-1}}$
Average daily flow at stream-gaging station B:	2.355 $\mathrm{m^3\ d^{-1}}$
Increase in flow due to ground-water discharge:	0.130 $\mathrm{m^3\ s^{-1}}$
Total *daily* ground-water discharge to stream:	11,232 $\mathrm{m^3\ d^{-1}}$
Discharge from half of aquifer (one side of the stream):	5,616 $\mathrm{m^3\ d^{-1}}$
Distance (x) between stations A and B:	5,000 m
Average thickness of aquifer (b):	50 m
Average slope of the water table (dh/dl) determined from measurements in the observation wells:	1 m/2,000 m

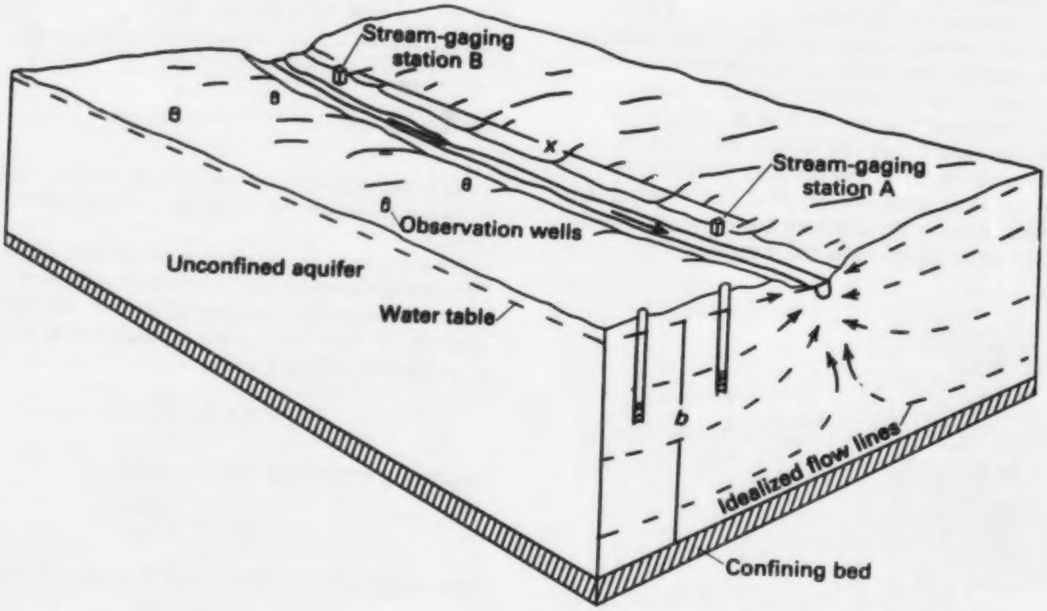


Figure 35. Hydrologic Situation Allowing Calculation of Transmissivity

By equation 19,

$$T = \frac{Q}{W} \times \frac{dl}{dh} = \frac{5{,}616\ \mathrm{m}^3}{\mathrm{d} \times 5{,}000\ \mathrm{m}} \times \frac{2{,}000\ \mathrm{m}}{1\ \mathrm{m}} = 2{,}246\ \mathrm{m}^2\,\mathrm{d}^{-1}$$

The hydraulic conductivity is determined from equation 15 as follows:

$$K = \frac{T}{b} = \frac{2{,}246\ \mathrm{m}^2}{\mathrm{d} \times 50\ \mathrm{m}} = 45\ \mathrm{m}\,\mathrm{d}^{-1}$$

Because transmissivity depends on both K and b, its value differs in different aquifers and from place to place in the same aquifer. Estimated values of transmissivity for the principal aquifers in different parts of the country range from less than 1 $\mathrm{m}^2\,\mathrm{d}^{-1}$ for some fractured sedimentary and igneous rocks to 100,000 $\mathrm{m}^2\,\mathrm{d}^{-1}$ for cavernous limestones and lava flows.

Finally, transmissivity replaces the term "coefficient of transmissibility" because, by convention, an aquifer is transmissive, and the water in it is transmissible.

Storage Coefficient

The abilities (capacities) of water-bearing materials to store and to transmit water are their most important hydraulic properties. Depending on the intended use of the information, these properties are given either in terms of a unit cube of the material or in terms of a unit prism of an aquifer.

Property	Unit cube of material	Unit prism of aquifer
Transmissive capacity	Hydraulic conductivity (K)	Transmissivity (T)
Available storage	Specific yield (S_y)	Storage coefficient (S)

The storage coefficient (S) is defined as the volume of water that an aquifer releases from or takes into storage per unit surface area of the aquifer per unit change in head. The storage coefficient is a dimensionless unit, as equation 20 shows, in which the units in the numerator and the denominator cancel:

$$S = \frac{\text{volume of water}}{\text{(unit area)(unit head change)}} = \frac{(\mathrm{m}^3)}{(\mathrm{m}^2)(\mathrm{m})} = \frac{\mathrm{m}^3}{\mathrm{m}^3} \quad (20)$$

The size of the storage coefficient depends on whether the aquifer is confined or unconfined, as Figure 36 shows. If the aquifer is confined, the water released from storage when the head declines comes from expansion of the water and from compression of the aquifer. Relative to a confined aquifer, the expansion of a given volume of water in response to a decline in pressure is very small. In a confined aquifer having a porosity of 0.2 and containing water at a temperature of about 15°C, expansion of the water alone releases about 3×10^{-7} m^3 of water per cubic meter of aquifer per meter of decline in head. To determine the storage coefficient of an aquifer due to expansion of the water, it is necessary to multiply the aquifer thickness by 3×10^{-7}. Thus, if only the expansion of water is considered, the storage coefficient of an aquifer 100 m thick would be 3×10^{-5}. The storage coefficient of most confined aquifers ranges from about 10^{-5} to 10^{-3} (0.00001 to 0.001). The difference between these values and the value due to expansion of the water is attributed to compression of the aquifer.

Figure 37 will aid in understanding this phenomenon. It shows a microscopic view of the contact between an aquifer and the overlying confining bed. The total load on the top of the aquifer is supported partly by the solid skeleton of the aquifer and partly by the hydraulic pressure exerted by the water in the aquifer. When the water pressure declines, more of the load must be sup-

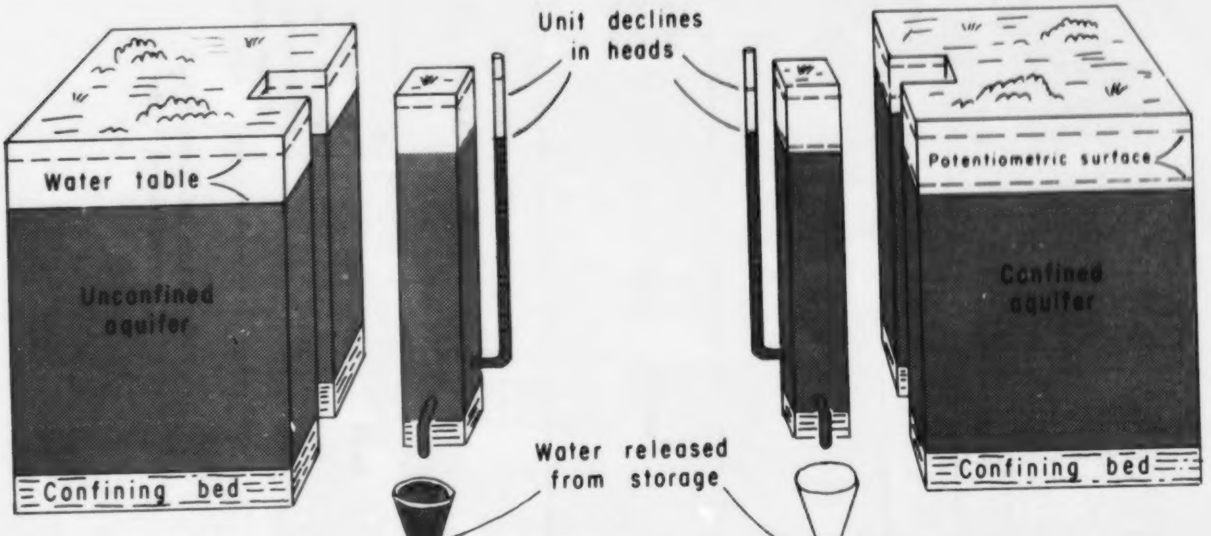


Figure 36. Size of the Storage Coefficient in Confined and Unconfined Aquifers

ported by the solid skeleton. As a result, the rock particles are distorted, and the pore space is reduced. The water forced from the pores when their volume is reduced represents the part of the storage coefficient due to compression of the aquifer.

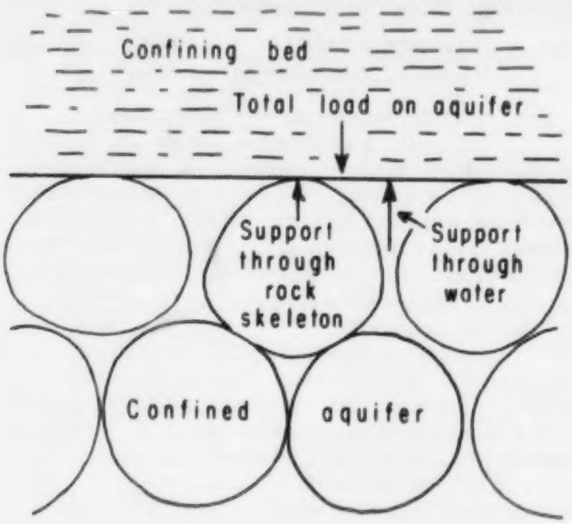


Figure 37. Microscopic View of the Contact Between an Aquifer and the Overlying Confining Bed

If the aquifer is unconfined, the predominant source of water is from gravity drainage of the sediments through which the decline in the water table occurs. In an unconfined aquifer, the volume of water derived from expansion of the water and compression of the aquifer is negligible. Thus, in such an aquifer, the storage coefficient is virtually equal to the specific yield and ranges from about 0.1 to about 0.3.

Because of the difference in the sources of storage, the storage coefficient of unconfined aquifers is 100 to 10,000 times the storage coefficient of confined aquifers, as Figure 36 illustrates. However, if water levels in an area are reduced to the point where an aquifer changes from a confined condition to an unconfined condition, the storage coefficient of the aquifer immediately increases from that of a confined aquifer to that of an unconfined aquifer.

Long-term withdrawals of water from many confined aquifers result in drainage of water both from clay layers within the aquifer and from adjacent confining beds. This drainage increases the load on the solid skeleton and results in compression of the aquifer and subsidence of the land surface. Subsidence of the land surface caused by drainage of clay layers has occurred in Arizona, California, Texas, and other areas.

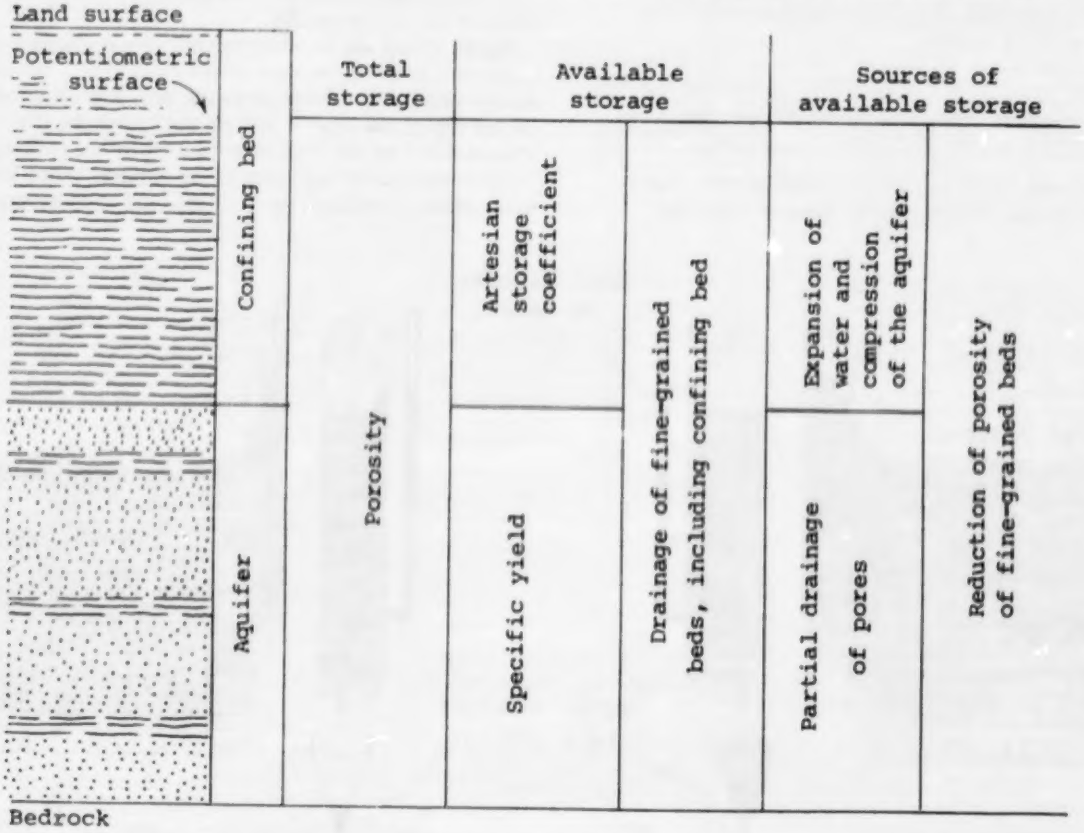

Land surface
Potentiometric surface
Confining bed
Aquifer
Bedrock
Total storage
Porosity
Available storage
Artesian storage coefficient
Specific yield
Drainage of fine-grained beds, including confining bed
Sources of available storage
Expansion of water and compression of the aquifer
Partial drainage of pores
Reduction of porosity of fine-grained beds

Figure 38. Potential Sources of Water in a Two-Unit Ground-Water System

The potential sources of water in a two-unit ground-water system consisting of a confining bed and a confined aquifer are shown in Figure 38. The figure is based on the assumption that water is removed in two separate stages—the first while the potentiometric surface is lowered to the top of the aquifer and the second by dewatering the aquifer.

The differences in the storage coefficients of confined and unconfined aquifers are of great importance in determining the response of the aquifers to stresses such as withdrawals through wells.

Cone of Depression

Both wells and springs serve as sources of ground-water supply. However, most springs having yields large enough to meet municipal, industrial, and large commercial and agricultural needs occur only in areas underlain by cavernous limestones and lava flows. Therefore, most ground-water needs are met by withdrawals from wells.

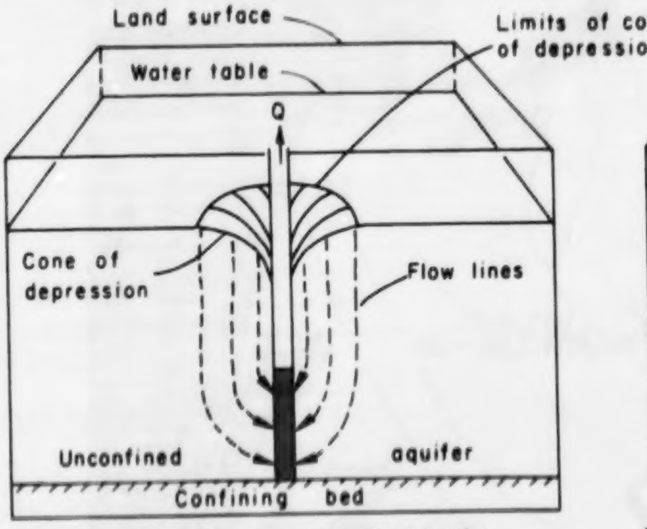


Figure 39. Withdrawals from an Unconfined Aquifer

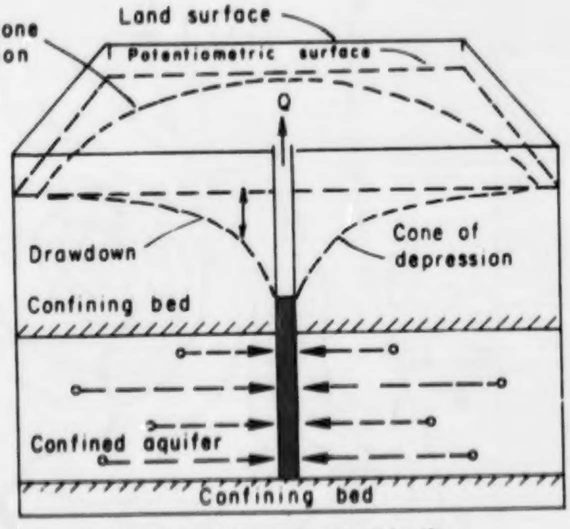


Figure 40. Withdrawals from a Confined Aquifer

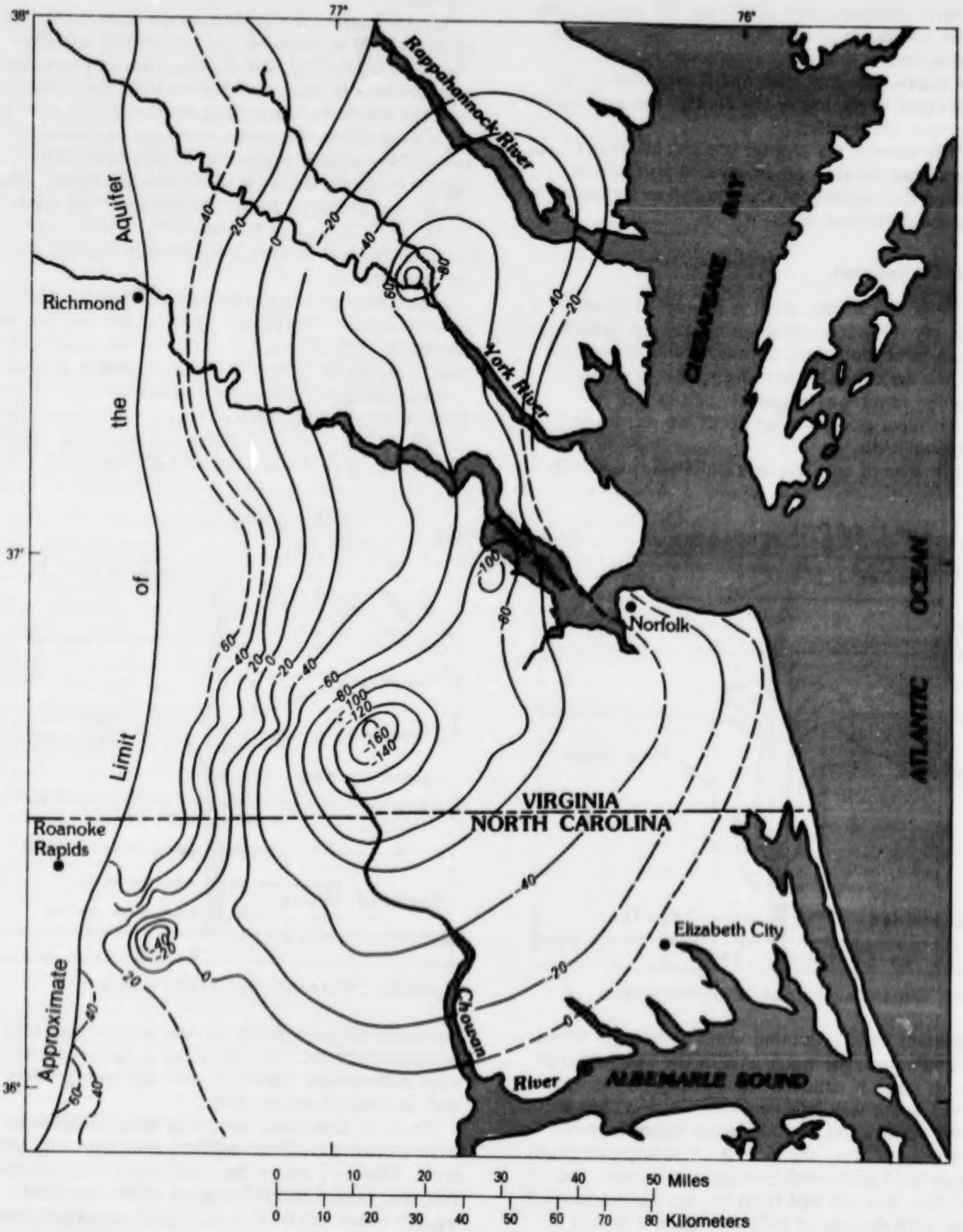
The response of aquifers to withdrawals from wells is an important topic in ground-water hydrology. When withdrawals start, the water level in the well begins to decline as water is removed from storage in the well. The head in the well falls below the level in the surrounding aquifer. As a result, water begins to move from the aquifer into the well. As pumping continues, the water level in the well continues to decline, and the rate of flow into the well from the aquifer continues to increase until the rate of inflow equals the rate of withdrawal.

Figures 39 and 40 illustrate that the movement of water from an aquifer into a well results in the formation of a *cone of depression*. Because water must converge on the well from all directions and because the area through which the flow occurs decreases toward the well, the hydraulic gradient must get steeper toward the well.

Several important differences exist between the cones of depression in confined and unconfined aquifers. Figure 39 shows that withdrawals from an unconfined aquifer result in drainage of water from the rocks through which the water table declines as the cone of depression forms. Because the storage coefficient of an unconfined aquifer equals the specific yield of the aquifer material, the cone of depression expands very slowly. On the other hand, dewatering of the aquifer results in a decrease in transmissivity, which causes, in turn, an increase in drawdown both in the well and in the aquifer.

Figure 40 shows that withdrawals from a confined aquifer cause a drawdown in artesian pressure but do not (normally) cause a dewatering of the aquifer. The water withdrawn from a confined aquifer is derived from expansion of the water and compression of the rock skeleton of the aquifer. (See the "Storage Coefficient" section of this chapter.) The very small storage coefficient of confined aquifers results in a very rapid expansion of the cone of depression. Consequently, the mutual interference of expanding cones around adjacent wells occurs more rapidly in confined aquifers than it does in unconfined aquifers.

Cones of depression caused by large withdrawals from extensive confined aquifers can affect very large areas. Figure 41 shows the overlapping cones of depression that existed in 1981 in an extensive confined aquifer composed of unconsolidated sands and interbedded silt and clay of Cretaceous age in the central part of the Atlantic Coastal Plain. The cones of depression are caused by withdrawals of about 277,000 m^3 d^{-1} (73,000,000 gal d^{-1}) from well fields in Virginia and North Carolina. (See the "Source of Water Derived From Wells" section of this chapter.)



Explanation

Water levels are in feet

National Geodetic Vertical Datum 1929

Figure 41. Potentiometric Surface of the Lowermost Cretaceous Aquifer in Southeastern Virginia and Northeastern North Carolina

Source of Water Derived from Wells

Both the economical development and the effective management of any ground-water system require an understanding of the response of the system to withdrawals from wells. The first concise description of the hydrologic principles involved in this response was presented by C. V. Theis in a paper published in 1940.

Theis pointed out that the response of an aquifer to withdrawals from wells depends on:

1. The rate of expansion of the cone of depression caused by the withdrawals, which depends on the transmissivity and the storage coefficient of the aquifer.
2. The distance to areas in which the rate of water discharging from the aquifer can be reduced.
3. The distance to recharge areas in which the rate of recharge can be increased.

Figure 42 shows that over a sufficiently long period of time under natural conditions—that is, before the start of withdrawals—the discharge from every ground-water system equals the recharge to it. In other words,

$$\text{natural discharge } (D) = \text{natural recharge } (R) \qquad (21)$$

In the eastern part of the United States and in the more humid areas in the West, the amount and distribution of precipitation are such that the period of time over which discharge and recharge balance may be less than a year or, at most, a few years. In the drier parts of the country—that is, in the areas that generally receive less than about 500 mm of precipitation annually — the period over which discharge and recharge balance may be several years or even centuries. Over shorter periods of time, differences between discharge and recharge involve changes in ground-water storage. In other words, when discharge exceeds recharge, ground-water storage (S) is reduced by an amount ΔS equal to the difference between discharge and recharge. Thus,

$$D = R + \Delta S \qquad (22)$$

Conversely, when recharge exceeds discharge, ground-water storage is increased. Thus,

$$D = R - \Delta S \qquad (23)$$

Figure 43 shows that when withdrawal through a well begins, water is removed from storage in its vicinity as the cone of depression develops. Thus, the withdrawal (Q) is balanced by a reduction in ground-water storage. In other words,

$$Q = \Delta S \qquad (24)$$

As the cone of depression expands outward from the pumping well, it may reach an area where water is discharging from the aquifer. Figure 44 shows that the hydraulic gradient will be reduced toward the discharge area, and the rate of natural discharge will decrease. To the extent that the decrease in natural discharge compensates for the pumpage, the rate at which water is being removed from storage will also decrease, and the rate of expansion of the cone of depression will decline. If and when the reduction in natural discharge (ΔD) equals the rate of withdrawal (Q), a new balance will be established in the aquifer. This balance in symbolic form is

$$(D - \Delta D) + Q = R \qquad (25)$$

Conversely, if the cone of depression expands into a recharge area rather than into a natural discharge area, the hydraulic gradient between the recharge area and the pumping well will be increased. If, under natural conditions, more water was available in the recharge area than the aquifer could accept (the condition that Theis referred to as one of *rejected recharge*), the increase in the gradient away from the recharge area will permit more recharge to occur, and the rate of growth of the cone of depression will decrease. If and when the increase in recharge (ΔR) equals the rate of withdrawal (Q), a new balance will be established in the aquifer, and expansion of the cone of depression will cease. The new balance in symbolic form is

$$D + Q = R + \Delta R \qquad (26)$$

In the eastern part of the United States, gaining streams are relatively closely spaced, and areas in which rejected recharge occurs are relatively unimportant. In this region, the growth of cones of depression first commonly causes a reduction in natural discharge. Figure 45 shows that, if the pumping wells are near a stream or if the withdrawals are continued long enough, ground-water discharge to a stream may be stopped entirely in the vicinity of the wells, and water may be induced to move from the stream into the aquifer. In other words, the tendency in this region is for withdrawals to change discharge areas into recharge areas. This consideration is important where the streams contain brackish or polluted water or where the streamflow is committed or required for other purposes.

To summarize, the withdrawal of ground water through a well reduces the water in storage in the source aquifer during the growth of the cone of depression. When and if the cone of depression ceases to expand, the rate of withdrawal is being balanced by a reduction in the rate of natural discharge and (or) by an increase in the rate of recharge. Under this condition,

$$Q = \Delta D + \Delta R \qquad (27)$$

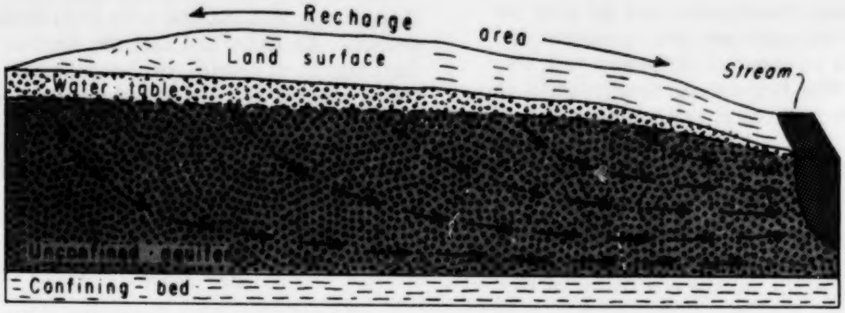


Figure 42. Discharge (*D*) = Recharge (*R*)

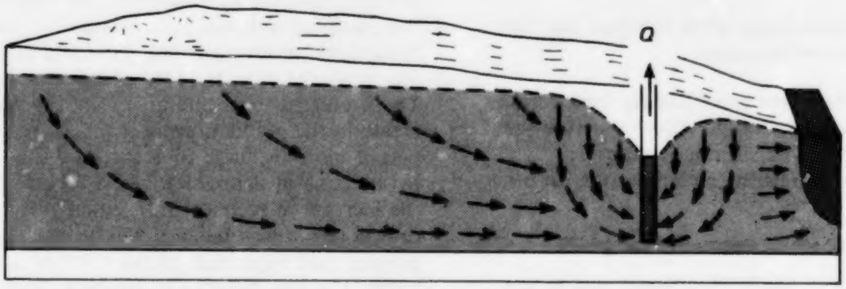


Figure 43. Withdrawal (*Q*) = Reduction in Storage (Δ*S*)

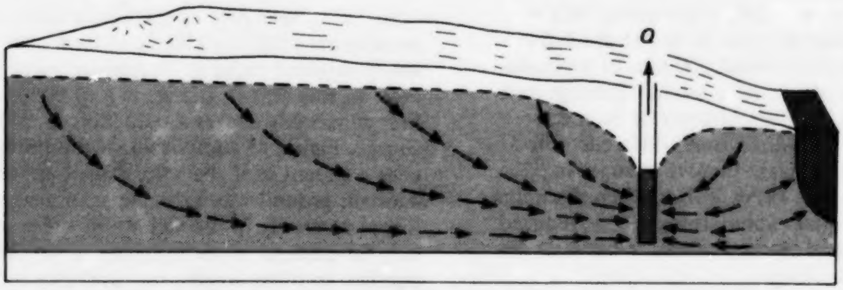


Figure 44. Withdrawal (*Q*) = Reduction in Storage (Δ*S*) + Reduction in Discharge (Δ*D*)

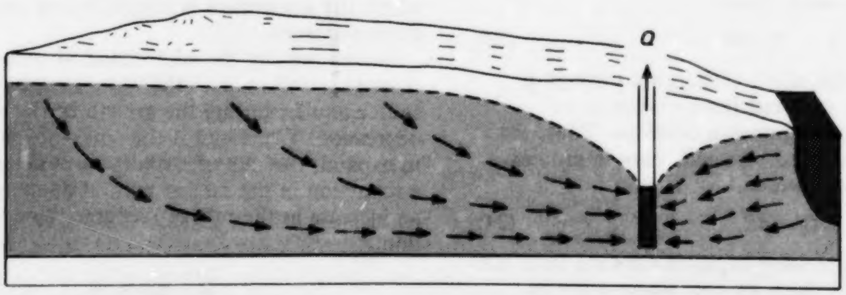


Figure 45. Withdrawal (*Q*) = Reduction in Discharge (Δ*D*) + Increase in Recharge (Δ*R*)

Aquifer Tests

Determining the yield of ground-water systems and evaluating the movement and fate of ground-water pollutants require, among other information, knowledge of:

1. The position and thickness of aquifers and confining beds.
2. The transmissivity and storage coefficient of the aquifers.
3. The hydraulic characteristics of the confining beds.
4. The position and nature of the aquifer boundaries.
5. The location and amounts of ground-water withdrawals.
6. The locations, kinds, and amounts of pollutants and pollutant practices.

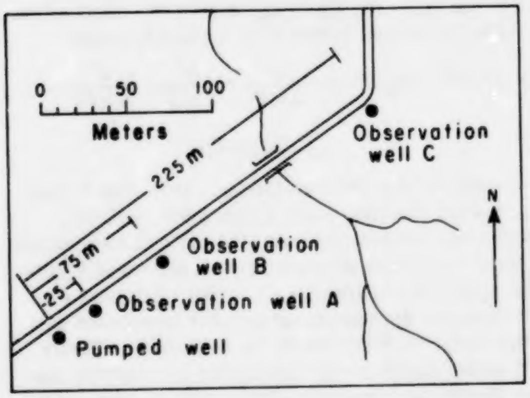


Figure 46. Map of Aquifer Test Site

Acquiring knowledge on these factors requires both geologic and hydrologic investigations. One of the most important hydrologic studies involves analyzing the change, with time, in water levels (or total heads) in an aquifer caused by withdrawals through wells. This type of study is referred to as an *aquifer test* and, in most cases, includes pumping a well at a constant rate for a period ranging from several hours to several days and measuring the change in water level in observation wells located at different distances from the pumped well. Figure 46 shows the map of an aquifer test site.

Successful aquifer tests require, among other things:

1. Determination of the prepumping water-level trend (that is, the regional trend).
2. A carefully controlled constant pumping rate.
3. Accurate water-level measurements made at precisely known times during both the drawdown and the recovery periods.

Drawdown is the difference between the water level at any time during the test and the position at which the water level would have been if withdrawals had not started. Drawdown is very rapid at first. As pumping continues and the cone of depression expands, as Figure 47 shows, the rate of drawdown decreases.

The *recovery* of the water level under ideal conditions is a mirror image of the drawdown. The change in water level during the recovery period is the same as if withdrawals had continued at the same rate from the pumped well but, at the moment of pump cutoff, a recharge well had begun recharging water at the same point and at the same rate. Therefore, the recovery of the water level is the difference between the actual measured level and the projected pumping level.

In addition to the constant-rate aquifer test mentioned above, analytical methods have also been developed for several other types of aquifer tests. These methods include tests in which the rate of withdrawal is variable and tests that involve leakage of water across confining beds into confined aquifers. The analytical methods available also permit analysis of tests conducted on both vertical wells and horizontal wells or drains.

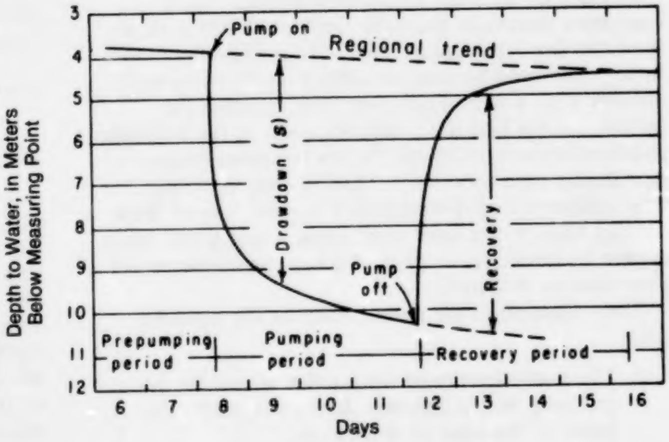


Figure 47. Change of Water Level in Well B

The most commonly used method of analysis of aquifer-test data—that for a vertical well pumped at a constant rate from an aquifer not affected by vertical leakage and lateral boundaries—will be covered in the discussion of "Analysis of Aquifer-Test Data." The method of analysis requires the use of a type curve based on the values of *W*(*u*) and 1/*u* listed in Table 5. Preparation and use of the type curve are covered in the following discussion.

Table 5. Selected Values of $W(u)$ for Values of $1/u$

$1/u$	10	7.69	5.88	5.00	4.00	3.33	2.86	2.5	2.22	2.00	1.67	1.43	1.25	1.11
10^{-1}	0.219	0.135	0.075	0.049	0.025	0.013	0.007	0.004	0.002	0.001	0.000	0.000	0.000	0.000
1	1.82	1.59	1.36	1.22	1.04	.91	.79	.70	.63	.56	.45	.37	.31	.26
10	4.04	3.78	3.51	3.35	3.14	2.96	2.81	2.68	2.57	2.47	2.30	2.15	2.03	1.92
10^2	6.33	6.07	5.80	5.64	5.42	5.23	5.08	4.95	4.83	4.73	4.54	4.39	4.26	4.14
10^3	8.63	8.37	8.10	7.94	7.72	7.53	7.38	7.25	7.13	7.02	6.84	6.69	6.55	6.44
10^4	10.94	10.67	10.41	10.24	10.02	9.84	9.68	9.55	9.43	9.33	9.14	8.99	8.86	8.74
10^5	13.24	12.98	12.71	12.55	12.32	12.14	11.99	11.85	11.73	11.63	11.45	11.29	11.16	11.04
10^6	15.54	15.28	15.01	14.85	14.62	14.44	14.29	14.15	14.04	13.93	13.75	13.60	13.46	13.34
10^7	17.84	17.58	17.31	17.15	16.93	16.74	16.59	16.46	16.34	16.23	16.05	15.90	15.76	15.65
10^8	20.15	19.88	19.62	19.45	19.23	19.05	18.89	18.76	18.64	18.54	18.35	18.20	18.07	17.95
10^9	22.45	22.19	21.92	21.76	21.53	21.35	21.20	21.06	20.94	20.84	20.66	20.50	20.37	20.25
10^{10}	24.75	24.49	24.22	24.06	23.83	23.65	23.50	23.36	23.25	23.14	22.96	22.81	22.67	22.55
10^{11}	27.05	26.79	26.52	26.36	26.14	25.96	25.80	25.67	25.55	25.44	25.26	25.11	24.97	24.86
10^{12}	29.36	20.09	28.83	28.66	28.44	28.26	28.10	27.97	27.85	27.75	27.56	27.41	27.28	27.16
10^{13}	31.66	31.40	31.13	30.97	30.74	30.56	30.41	30.27	30.15	30.05	29.87	29.71	29.58	29.46
10^{14}	33.96	33.70	33.43	33.27	33.05	32.86	32.71	32.58	32.46	32.35	32.17	32.02	31.88	31.76

Examples: when $1/u = 10 \times 10^{-1}$, $W(u) = 0.219$; when $1/u = 3.33 \times 10^2$, $W(u) = 5.23$.

Analysis of Aquifer-Test Data

In 1935, C. V. Theis of the New Mexico Water Resources District of the U.S. Geological Survey developed the first equation to include time of pumping as a factor that could be used to analyze the effect of withdrawals from a well. Thus, the *Theis equation* permitted, for the first time, determination of the hydraulic characteristics of an aquifer before the development of new steady-state conditions resulting from pumping. The importance of this capability may be realized from the fact that, under most conditions, a new steady state cannot be developed or that, if it can, many months or years may be required.

Theis assumed in the development of the equation that:

1. The transmissivity of the aquifer tapped by the pumping well is constant during the test to the limits of the cone of depression.
2. The water withdrawn from the aquifer is derived entirely from storage and is discharged instantaneously with the decline in head.
3. The discharging well penetrates the entire thickness of the aquifer, and its diameter is small in comparison with the pumping rate, so that storage in the well is negligible.

These assumptions are most nearly met by confined aquifers at sites remote from their boundaries. However, if certain precautions are observed, the equation can also be used to analyze tests of unconfined aquifers.

The forms of the Theis equation used to determine the transmissivity and storage coefficient are

$$T = \frac{Q\,W(u)}{4\pi s} \tag{28}$$

$$S = \frac{4Ttu}{r^2} \tag{29}$$

where T is transmissivity, S is the storage coefficient, Q is the pumping rate, s is drawdown, t is time, r is the distance from the pumping well to the observation well, $W(u)$ is the well function of u, which equals

$$-0.577216 - \log_e u + u - \frac{u^2}{2 \times 2!} + \frac{u^3}{3 \times 3!} - \frac{u^4}{4 \times 4!} + \ldots$$

and $u = (r^2 S)/(4Tt)$.

The form of the Theis equation is such that it cannot be solved directly. To overcome this problem, Theis devised a convenient graphic method of solution that involves the use of a type curve, shown in Figure 48. To apply this method, a data plot of drawdown versus time (or drawdown versus t/r^2) is matched to the type curve of $W(u)$ versus $1/u$ as shown in Figure 49. At some convenient point on the overlapping part of the sheets containing the data plot and type curve, values of s, t (or t/r^2), $W(u)$, and $1/u$ are noted. These values are then substituted in equations 28 and 29, which are solved for T and S, respectively.

A Theis type curve of $W(u)$ versus $1/u$ can be prepared from the values given in the table contained in the preceding section, "Aquifer Tests." The data points are plotted on logarithmic graph paper—that is, graph paper having logarithmic divisions in both the x and y directions.

The dimensional units of transmissivity (T) are L^2t^{-1}, where L is length and t is time in days. Thus, if Q in equation 28 is in cubic meters per day and s is in meters, T will be in square meters per day. Similarly, if, in equation 29, T is in square meters per day, t is in days, and r is in meters, S will be dimensionless.

Traditionally, in the United States, T has been expressed in units of gallons per day per foot. The common practice now is to report transmissivity in units of square meters per day or square feet per day. If Q is measured in gallons per minute, as is still normally the case, and drawdown is measured in feet, as is also nor-

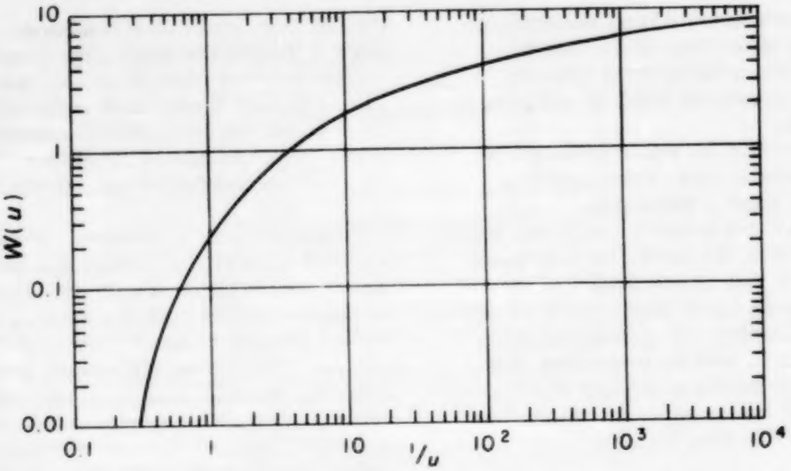


Figure 48. Theis Type Curve

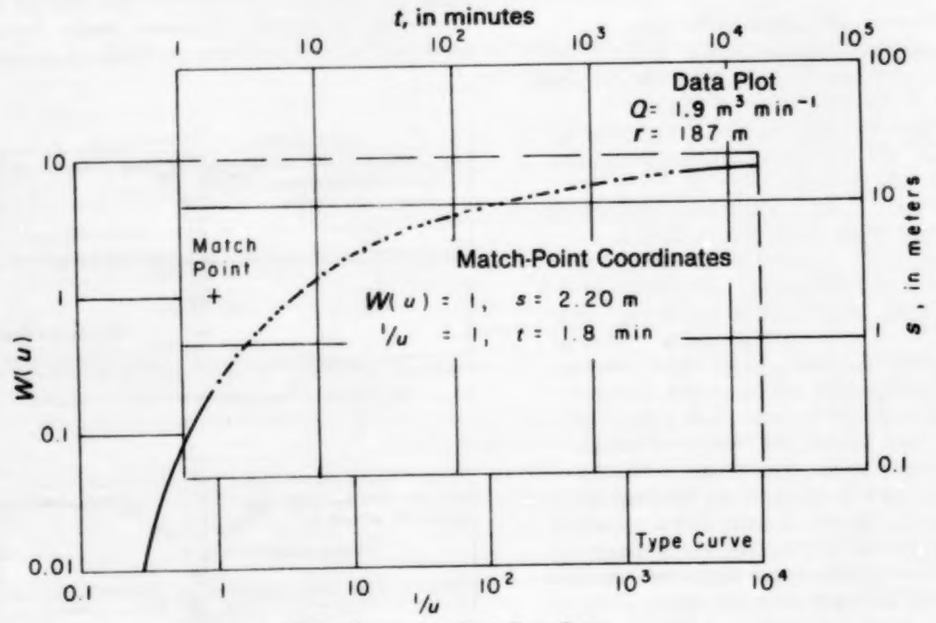


Figure 49. Data Plot of Drawdown Versus Time Matched to Theis Type Curve

mally the case, equation 28 is modified to obtain T in square feet per day as follows:

$$T = \frac{Q\,W(u)}{4\pi s} = \frac{\text{gal}}{\text{min}} \times \frac{1{,}440\ \text{min}}{\text{d}} \times \frac{\text{ft}^3}{7.48\ \text{gal}} \times \frac{1}{\text{ft}} \times \frac{W(u)}{4\pi}$$

or

$$T(\text{in ft}^2\,\text{d}^{-1}) = \frac{15.3Q\,W(u)}{s}$$

(when Q is in gallons per minute and s is in feet). To convert square feet per day to square meters per day, divide by 10.76.

The storage coefficient is dimensionless. Therefore, if T is in square feet per day, t is in minutes, and r is in feet, then, by equation 29,

$$S = \frac{4Ttu}{r^2} = \frac{4}{1} \times \frac{\text{ft}^2}{\text{d}} \times \frac{\text{min}}{\text{ft}^2} \times \frac{\text{d}}{1{,}440\ \text{min}}$$

or

$$S = \frac{Ttu}{360\,r^2}$$

(When T is in square feet per day, t is in minutes, and r is in feet).

Analysis of aquifer-test data using the Theis equation involves plotting both the type curve and the test data on logarithmic graph paper. If the aquifer and the conditions of the test satisfy Theis's assumptions, the type curve has the same shape as the cone of depression along any line radiating away from the pumping well and the drawdown graph at any point in the cone of depression.

Use of the Theis equation for unconfined aquifers involves two considerations. First, if the aquifer is relatively fine grained, water is released slowly over a period of hours or days, not instantaneously with the decline in head. Therefore, the value of S determined from a short-period test may be too small.

Second, if the pumping rate is large and the observation well is near the pumping well, dewatering of the aquifer may be significant, and the assumption that the transmissivity of the aquifer is constant is not satisfied. The effect of dewatering of the aquifer can be eliminated with the following equation:

$$s' = s - \left(\frac{s^2}{2b}\right) \qquad (30)$$

where s is the observed drawdown in the unconfined aquifer, b is the aquifer thickness, and s' is the drawdown that would have occurred if the aquifer had been confined (that is, if no dewatering had occurred).

To determine the transmissivity and storage coefficient of an unconfined aquifer, a data plot consisting of s' versus t (or t/r^2) is mathed with the Theis type curve of $W(u)$ versus $1/u$. Both s and b in equation 30 must be in the same units, either feet or meters.

As noted above, Theis assumed in the development of his equation that the discharging well penetrates the entire thickness of the aquifer. However, because it is not always possible, or necessarily desirable, to design a well that fully penetrates the aquifer under development, most discharging wells are open to only a part of the aquifer that they draw from. Such *partial penetration* creates vertical flow in the vicinity of the discharging well that may affect drawdowns in observation wells located relatively close to the discharging well. Drawdowns in observation wells that are open to the same zone as the discharging well will be larger than the drawdowns in wells at the same distance from the discharging well but open to other zones. The possible effect of partial penetration on drawdowns must be considered in the analysis of aquifer-test data. If aquifer-boundary and other conditions permit, the problem can be avoided by locating observation wells beyond the zone in which vertical flow exists.

Time-Drawdown Analysis

The Theis equation is only one of several methods that have been developed for the analysis of aquifer-test data. (See the "Analysis of Aquifer-Test Data" section of this chapter.) Another method, and one that is somewhat more convenient to use, was developed by C. E. Jacob from the Theis equation. The greater convenience of the Jacob method derives partly from its use of semilogarithmic graph paper instead of the logarithmic paper used in the Theis method and from the fact that, under ideal conditions, the data plot along a straight line rather than along a curve.

However, it is essential to note that, whereas the Theis equation applies at all times and places (if the assumptions are met), Jacob's method applies only under certain additional conditions. These conditions must also be satisfied in order to obtain reliable answers.

To understand the limitations of Jacob's method, we must consider the changes that occur in the cone of depression during an aquifer test. The changes that are of concern involve both the *shape of the cone* and the *rate of drawdown*. As the cone of depression migrates outward from a pumping well, its shape (and, therefore, the hydraulic gradient at different points in the cone) changes. We can refer to this condition as *unsteady shape*. At the start of withdrawals Figure 50 shows that the entire cone of depression has an unsteady shape. After a test has been underway for some time, Figure 51 shows that the cone of depression begins to assume a relatively *steady shape*, first at the pumping well and then gradually to greater and

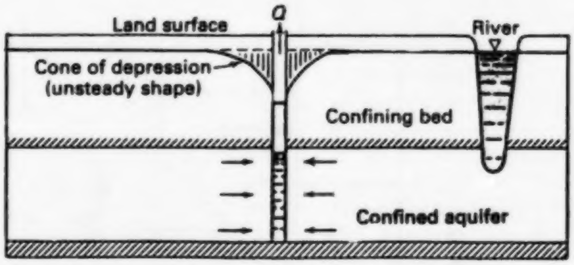


Figure 50. Cone of Depression at Start of Withdrawal

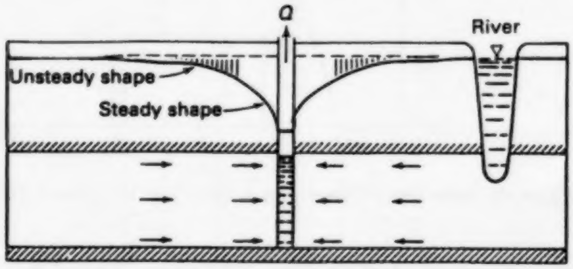


Figure 51. Cone of Depression During Test

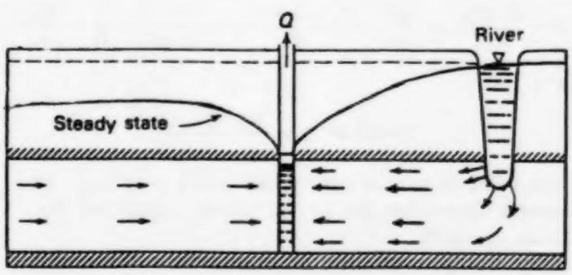


Figure 52. Steady-State Cone of Depression

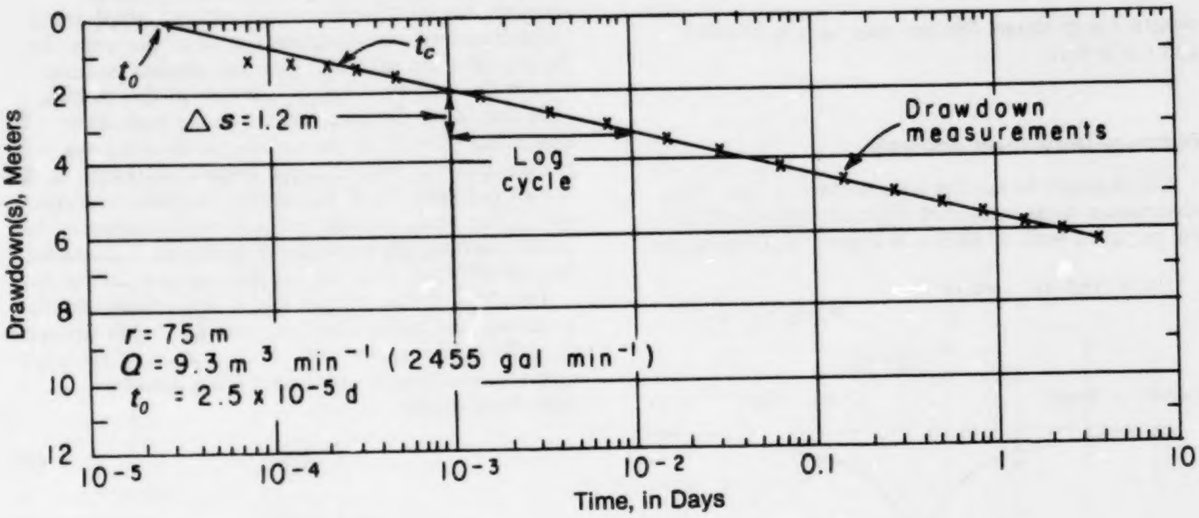


Figure 53. Time-Drawdown Graph

greater distances. If withdrawals continue long enough for increases in recharge and (or) reductions in discharge to balance the rate of withdrawal, drawdowns cease, and the cone of depression is said to be in a *steady state,* as Figure 52 shows.

The Jacob method is applicable only to the zone in which steady-shape conditions prevail or to the entire cone only after steady-state conditions have developed. For practical purposes, this condition is met when $u = (r^2S)/)4Tt)$ is equal to or less than about 0.05. Substituting this value in the equation for u and solving for t, we can determine the time at which steady-shape conditions develop at the outermost observation well. Thus,

$$t_c = \frac{7{,}200\, r^2 S}{T} \tag{31}$$

where t_c is the time, in minutes, at which steady-shape conditions develop, r is the distance from the pumping well, in feet (or meters), S is the estimated storage coefficient (dimensionless), and T is the estimated transmissivity, in square feet per day (or square meters per day).

After steady-shape conditions have developed, the drawdowns at an observation well begin to fall along a straight line on semilogarithmic graph paper, as Figure 53 shows. Before that time, the drawdowns plot below the extension of the straight line. When a time-drawdown graph is prepared, drawdowns are plotted on the vertical (arithmetic) axis versus time on the horizontal (logarithmic) axis.

The slope of the straight line is proportional to the pumping rate and to the transmissivity. Jacob derived the following equations for determination of transmissivity and storage co-efficient from the time-drawdown graphs:

$$T = \frac{2.3\,Q}{4\pi\Delta s} \tag{32}$$

$$S = \frac{2.25\,Tt_o}{r^2} \tag{33}$$

where Q is the pumping rate, Δs is the drawdown across one log cycle, t_o is the time at the point where the straight line intersects the zero-drawndown line, and r is the distance from the pumping well to the observation well.

Equations 32 and 33 are in consistent units. Thus, if Q is in cubic meters per day and s is in meters, T is in square meters per day. S is dimensionless, so that, in equation 33, if T is in square meters per day, then r must be in meters and t_o must be in days.

It is still common practice in the United States to express Q in gallons per minute, s in feet, t in minutes, r in feet, and T in square feet per day. We can modify equations 32 and 33 for direct substitution of these units as follows:

$$T = \frac{2.3\,Q}{4\pi\Delta s} = \frac{2.3}{4\pi} \times \frac{\text{gal}}{\text{min}} \times \frac{1{,}440\ \text{min}}{\text{d}} \times \frac{\text{ft}^2}{7.48\ \text{gal}} \times \frac{1}{\text{ft}}$$

$$T = \frac{35\,Q}{s} \tag{34}$$

(where T is in square feet per day, Q is in gallons per minute, and s is in feet) and

$$S = \frac{2.25\,Tt_o}{r^2} = \frac{2.25}{1} \times \frac{\text{ft}^2}{\text{d}} \times \frac{\text{min}}{\text{ft}^2} \times \frac{\text{d}}{1{,}440\ \text{min}}$$

$$S = \frac{Tt_o}{640\,r^2} \qquad (35)$$

(where T is in square feet per day, t_o is in minutes, and r is in feet).

Distance-Drawdown Analysis

It is desirable in aquifer tests to have at least three observation wells located at different distances from the pumping well, as shown in Figure 54. Drawdowns measured at the same time in these wells can be analyzed with the Theis equation and type curve to determine the aquifer transmissivity and storage coefficient.

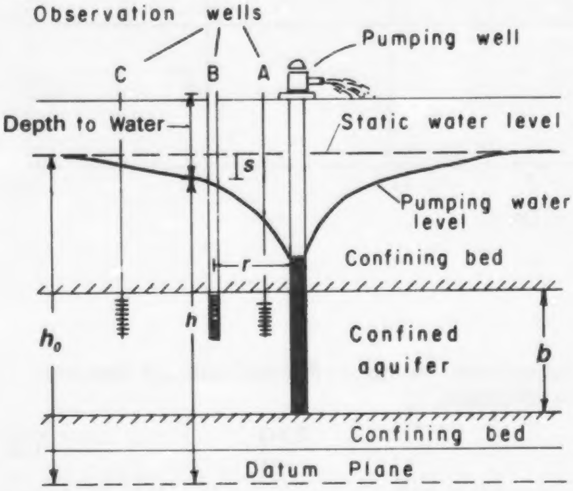


Figure 54. Desirable Location for Observation Wells in Aquifer Tests.

After the test has been underway long enough, drawdowns in the wells can also be analyzed by the Jacob method, either through the use of a time-drawdown graph using data from individual wells or through the use of a distance-drawdown graph using "simultaneous" measurements in all of the wells. To determine when sufficient time has elapsed, see the "Time-Drawdown Analysis" section of this chapter.

In the Jacob distance-drawdown method, drawdowns are plotted on the vertical (arithmetic) axis versus distance on the horizontal (logarithmic) axis, as shown in Figure 55. If the aquifer and test conditions satisfy the Theis assumptions and the limitation of the Jacob method, the drawdowns measured at the same time in different wells should plot along a straight line.

The slope of the straight line is proportional to the pumping rate and to the transmissivity. Jacob derived the following equations for determination of the transmissivity and storage coefficient from distance-drawdown graphs:

$$T = \frac{2.3\,Q}{2\pi \Delta s} \qquad (36)$$

$$S = \frac{2.25\,Tt}{r_o^2} \qquad (37)$$

where Q is the pumping rate, Δs is the drawdown across one log cycle, t is the time at which the drawdowns were measured, and r_o is the distance from the pumping well to the point where the straight line intersects the zero-drawdown line.

Equations 36 and 37 are in consistent units. For the inconsistent units still in relatively common use in the United States, equations 36 and 37 should be used in the following forms:

$$T = \frac{70\,Q}{\Delta s} \qquad (38)$$

(where T is in square feet per day, Q is in gallons per minute, and s is in feet) and

$$S = \frac{Tt}{640\,r_o^2} \qquad (39)$$

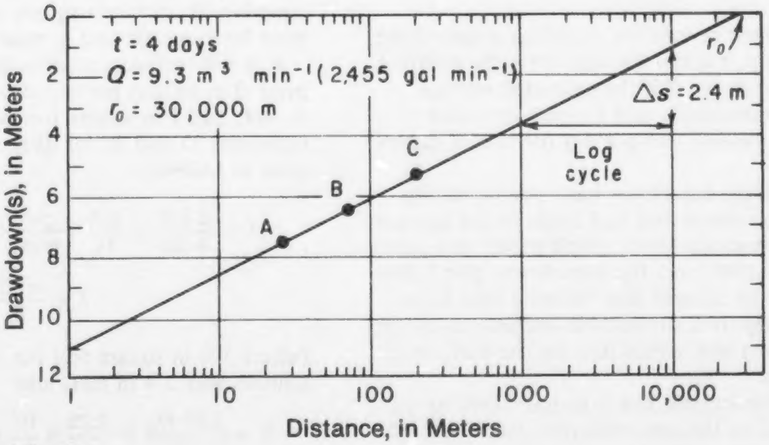


Figure 55. Distance-Drawdown Graph

(where T is in square feet per day, t is in minutes, and r_o is in feet).

The distance r_o does not indicate the outer limit of the cone of depression. Because nonsteady-shape conditions exist in the outer part of the cone, before the development of steady-state conditions, the Jacob method does not apply to that part. If the Theis equation were used to calculate drawdowns in the outer part of the cone, it would be found that they would plot below the straight line. In other words, the measurable limit of the cone of depression is beyond the distance r_o.

If the straight line of the distance-drawdown graph is extended inward to the radius of the pumping well, the drawdown indicated at that point is the drawdown in the aquifer outside of the well. If the drawdown inside the well is found to be greater than the drawdown outside, the difference is attributable to *well loss.* (See the "Single-Well Tests" section of this chapter.)

As noted in the section on "Hydraulic Conductivity," the hydraulic conductivities and, therefore, the transmissivities of aquifers may be different in different directions. These differences may cause drawdowns measured at the same time in observation wells located at the same distances but in different directions from the discharging well to be different. Where this condition exists, the distance-drawdown method may yield satisfactory results only where three or more observation wells are located in the same direction but at different distances from the discharging well.

Single-Well Tests

The most useful aquifer tests are those that include water-level measurements in observation wells. Such tests are commonly referred to as *multiple-well tests.* It is also possible to obtain useful data from production wells, even where observation wells are not available. Such tests are referred to as *single-well tests* and may consist of pumping a well at a single constant rate, or at two or more different but constant rates (see the "Well-Acceptance Tests and Well Efficiency" section of this chapter) or, if the well is not equipped with a pump, by "instantaneously" introducing a known volume of water into the well. This discussion will be limited to tests involving a single constant rate.

In order to analyze the data, it is necessary to understand the nature of the drawdown in a pumping well. Figure 56 shows that the total drawdown (s_t) in most, if not all, pumping wells consists of two components. One is the drawdown (s_a) in the aquifer, and the other is the drawdown (s_w) that occurs as water moves from the aquifer into the well and up the well bore to the pump intake. Thus, the drawdown in most pumping wells is greater than the drawdown in the aquifer at the radius of the pumping well.

The total drawdown (s_t) in a pumping well can be expressed in the form of the following equations:

$$s_t = s_a + s_w \tag{40}$$

$$s_t = BQ + CQ^2 \tag{41}$$

where s_a is the drawdown in the aquifer at the effective radius of the pumping well, s_w is well loss, Q is the pumping rate, B is a factor related to the hydraulic characteristics of the aquifer and the length of the pumping period, and C is a factor related to the characteristics of the well.

The factor C in equation 40 normally considered to be constant, so that, in a constant rate test, CQ^2 is also constant. As a result, the well loss (s_w) increases the total drawdown in the pumping well but does not affect the rate of change in the drawdown with time. It is, therefore, possible to analyze drawdowns in the pumping well with the Jacob time-drawdown method using

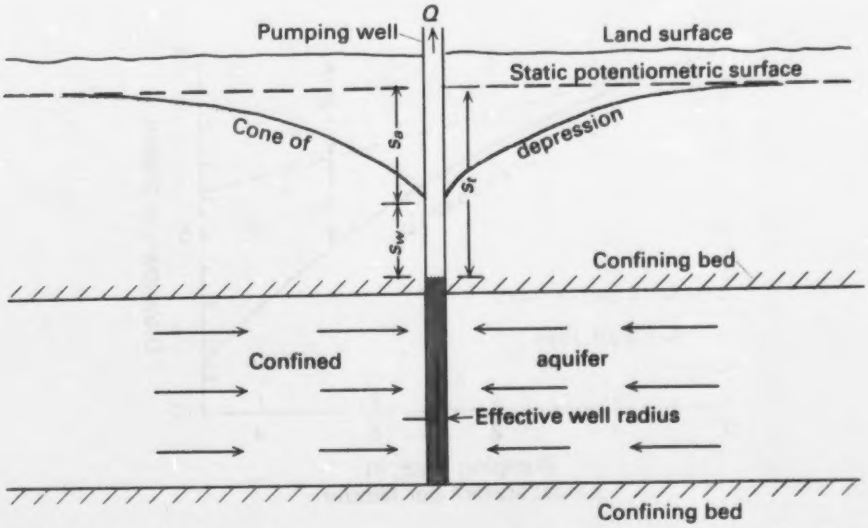


Figure 56. Two Components of Total Drawdown.

semilogarithmic graph paper. (See the "Time-Drawdown Analysis" section of this chapter.) As Figure 57 shows, drawdowns are plotted on the arithmetic scale versus time on the logarithmic scale and transmissivity is determined from the slope of the straight line through the use of the following equation:

$$T = \frac{2.3\,Q}{4\pi \Delta s} \qquad (32)$$

Where well loss is present in the pumping well, the storage coefficient cannot be determined by extending the straight line to the line of zero drawdown. Even where well loss is not present, the determination of the storage coefficient from drawdowns in a pumping well likely will be subject to large error because the effective radius of the well may differ significantly from the "nominal" radius.

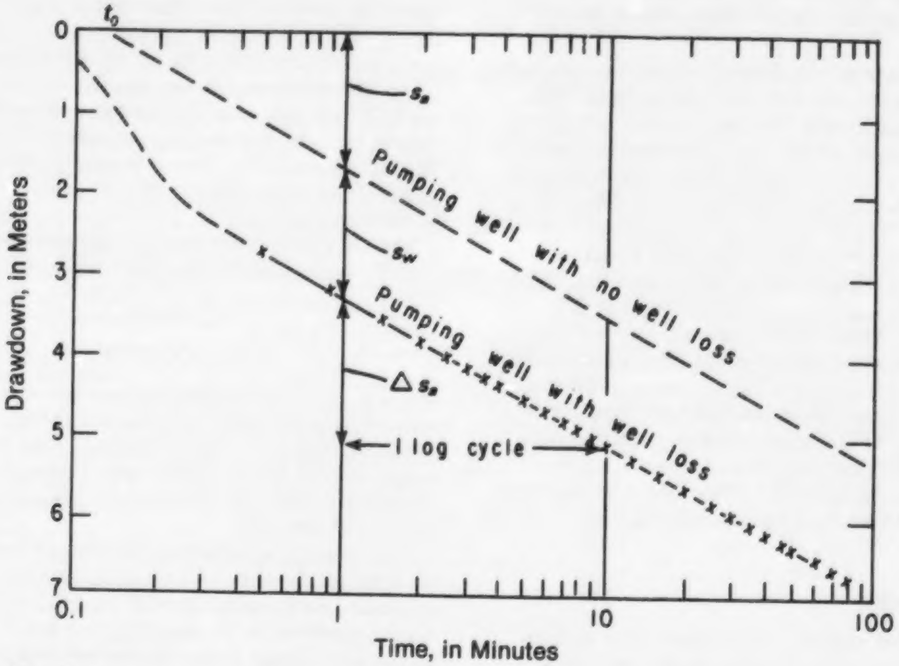


Figure 57. Drawdown Plot

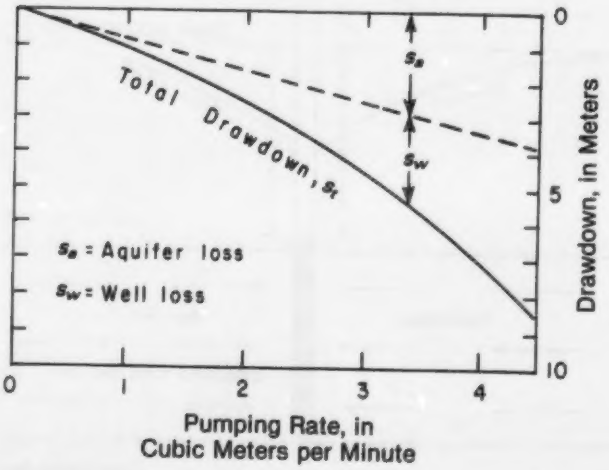


Figure 58. Relation of Pumping Rate and Drawdown

In equation 41, drawdown in the pumping well is proportional to the pumping rate. The factor B in the aquifer-loss term (BQ) increases with time of pumping as long as water is being derived from storage in the aquifer. The factor C in the well-loss term (CQ^2) is a constant if the characteristics of the well remain unchanged, but, because the pumping rate in the well-loss term is squared, drawdown due to well loss increases rapidly as the pumping rate is increased. The relation between pumping rates and drawdown in a pumping well, if the well was pumped for the same length of time at each rate, is shown in Figure 58. The effect of well loss on drawdown in the pumping well is important both in the analysis of data from pumping wells and in the design of supply wells.

Well-Acceptance Tests and Well Efficiency

Many supply-well contracts require a "guaranteed" yield, and some stipulate that the well reach a certain level of "efficiency." Most contracts also specify the length of the "drawdown test" that must be conducted to demonstrate that the yield requirement is met. For example, many States require that tests of public-supply wells be at least 24 hours. Tests of most industrial and irrigation wells probably do not exceed about 8 hours.

Well-acceptance tests, if properly conducted, not only can confirm the yield of a well and the size of the production pump that is needed but can also provide information of great value in well operation and maintenance. Such tests should, therefore, be conducted with the same care as aquifer tests made to determine the hydraulic characteristics of aquifers. A properly conducted test will include:

1. Determination of well interference from nearby pumping wells, based on accurate water-level measurements made before the drawdown test.
2. A pumping rate that is either held constant during the entire test, as shown in Figure 59(a), or increased in steps of equal length, as shown in Figure 59(b). The pumping rate during each step should be held constant, and the length of each step should be at least 2 hours.

Of these requirements, the constant, carefully regulated pumping rate or rates and the accurate water-level measurements are the most important. When a constant-rate well-acceptance test has been completed, the drawdown data can be analyzed to determine the aquifer transmissivity. (See the "Single-Well Tests" section of this chapter.)

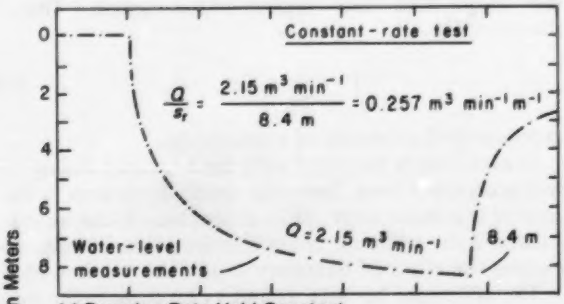
Many well-acceptance tests are made with temporary pump installations, usually powered with a gasoline or diesel engine. Instead of maintaining a constant rate for the duration of the test, the engine is frequently stopped to add fuel or to check the oil level or for numerous other reasons. The rate may also be increased and decreased on an irregular, unplanned schedule or, more commonly, gradually reduced during the test in an effort to maintain a pumping level above the pump intake. In such tests, the "yield" of the well is normally reported to be the final pumping rate.

Determining the long-term yield of a well from data collected during a short-period well-acceptance test is one of the most important, practical problems in ground-water hydrology. Two of the most important factors that must be considered are the extent to which the yield will decrease if the well is pumped continuously for periods longer than the test period and the effect on the yield of changes in the static (regional) water level from that existing at the time of the test.

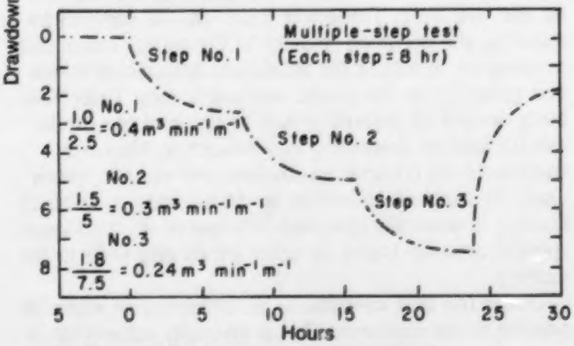
When data are available only from the production well and when the pumping rate was not held constant during the acceptance test, the estimate of the long-term yield must usually be based on an analysis of specific-capacity data. *Specific capacity* is the yield per unit of drawdown and is determined by dividing the pumping rate at any time during the test by the drawdown at the same time. Thus,

$$\text{specific capacity} = \frac{\text{pumping rate}}{\text{drawdown}} = \frac{Q}{s_t} \qquad (42)$$

Before the development of steady-state conditions, a part of the water pumped from an aquifer is derived from storage. The time required for steady-state conditions to develop depends largely on the distance to and characteristics of the recharge and discharge areas and the hydraulic characteristics of the aquifer. The time required to reach a steady state is independent of the pumping rate. At some places in some aquifers, a



(a) Pumping Rate Held Constant



(b) Pumping Rate Increased in Equal Steps

Figure 59. Pumping Rates for Well-Acceptance Tests

steady-state condition will be reached in several days, whereas, in others, six months to a year may be required; in some arid areas, a steady-state condition may never be achieved. Depending on the length of the well-acceptance test and the period required to reach a steady-state condition, it may be appropriate, in estimating the long-term yield of a well, to use a specific capacity smaller than that determined during the test.

Figure 60 shows the decline in a specific capacity with time when a well is pumped continuously at a constant rate and all the water is derived from storage in an isotropic and homogeneous acquifer. For convenience in preparing Figure 60, a value of 100 percent was assigned to the specific capacity 1 hour after the pump was started. The rate at which the specific capacity decreases depends on the decline of the water level due to depletion of storage and on the hydraulic characteristics of the aquifer. Differences in the rate for different aquifers are indicated by the width of the band on Figure 60. When withdrawals are derived entirely from storage, the specific capacity will decrease about 40 percent during the first year.

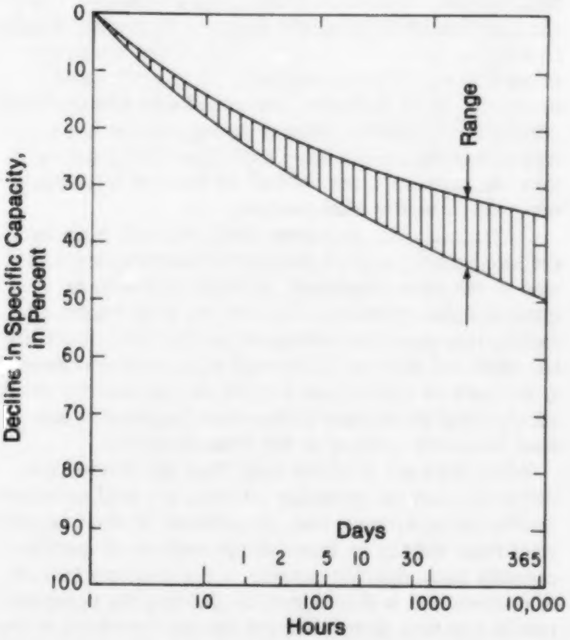


Figure 60. Decline in Specific Capacity With Time at a Continuous Pumping Rate

In predicting the long-term yield of a well, it is also necessary to consider changes in the static water level resulting from seasonal and long-term variations in recharge and declines due to other withdrawals from the aquifer. The long-term yield is equal to the specific capacity, determined from the well-acceptance test, and reduced as necessary to compensate for the long-term decline discussed in the above paragraph, multiplied by the available drawdown.

The *available drawdown* at the time of a well-acceptance test is equal to the difference between the static water level at that time and the lowest pumping level that can be imposed on the well. The lowest pumping level in a screened well is normally considered to be a meter or two above the top of the screen. In an unscreened (open-hole) well, it may be at the level of either the highest or the lowest water-bearing opening penetrated by the well. The choice of the highest or the lowest opening depends on the chemical composition of the water and whether water cascading from openings above the pumping level results in precipitation of minerals on the side of the well and on the pump intake. If such precipitation is expected, the maximum pumping level should not be below the highest opening. The yield of a well is not increased by a pumping level below the lowest opening, and the maximum yield may, in fact, be attained at a much higher level.

To predict the maximum continuous long-term yield, it is necessary to estimate how much the static water level, and thus the available drawdown, may decline from the position that it occupied during the acceptance test. Records of water-level fluctuations in long-term observation wells in the area will be useful in this effort.

Well efficiency is an important consideration both in well design and in well construction and development. The objective, of course, is to avoid excessive energy costs by designing and constructing wells that will yield the required water with the least drawdown.

Well efficiency can be defined as the ratio of the drawdown (s_a) in the aquifer at the radius of the pumping well to the drawdown (s_t) inside the well. (See the "Single-Well Tests" section of this chapter.) Thus, the equation

$$E = \frac{s_a}{s_t} \times 100 \qquad (43)$$

expresses well efficiency as a percentage.

Drawdowns in pumping wells are measured during well-acceptance tests. Determining the drawdown in the aquifer is a much more difficult problem. It can be calculated if the hydraulic characteristics of the aquifer, including the effect of boundary conditions, are known.

The difference between s_t and s_a is attributed to head losses as water moves from an aquifer into a well and up the well bore. These well losses can be reduced by reducing the entrance velocity of the water, which can be done by installing the maximum amount of screen and pumping at the lowest acceptable rate. Tests have been devised to determine well losses, and the results can be used to determine well efficiency. However, these tests are difficult to conduct and are not widely used. Because of difficulties in determining s_a, well efficiency is generally specified in terms of an "optimum" specific capacity based on other producing wells in the vicinity.

Under the best conditions, an efficiency of about 80 percent is the maximum that is normally achievable in most screened wells. Under less than ideal conditions, an efficiency of 60 percent is probably more realistic.

Specific Capacity and Transmissivity

As Figure 61 shows, the specific capacity of a well depends both on the hydraulic characteristics of the aquifer and on the construction and other features of the well. Values of specific capacity, available for many supply wells for which aquifer-test data are not available, are widely used by hydrologists to estimate transmissivity. Such estimates are used to evaluate regional differences in transmissivity and to prepare transmissivity maps for use in models of ground-water systems.

The factors that affect specific capacity include:

1. The transmissivity of the zone supplying water to the well, which, depending on the length of the screen or open hole, may be considerably less than the transmissivity of the aquifer.
2. The storage coefficient of the aquifer.
3. The length of the pumping period.
4. The effective radius of the well, which may be significantly greater than the "nominal" radius.
5. The pumping rate.

The Theis equation can be used to evaluate the effect of the first four factors on specific capacity. The last factor, pumping rate, affects the well loss and can be determined only from a stepped-rate test or an aquifer test in which drawdowns are measured in both the pumping well and observation wells.

The Theis equation, modified for the determination of transmissivity from specific capacity, is

$$T = \frac{W(u)}{4\pi} \times \frac{Q}{s} \qquad (44)$$

where T is transmissivity, Q/s is specific capacity, Q is the pumping rate, s is the drawdown, and $W(u)$ is the well function of u, where

$$u = \frac{r^2S}{4Tt} \qquad (45)$$

where r is the effective radius of the well, S is the storage coefficient, and t is the length of the pumping period preceding the determination of specific capacity.

For convenience in using equation 44, it is desirable to express $W(u)/4\pi$ as a constant. To do so, it is first

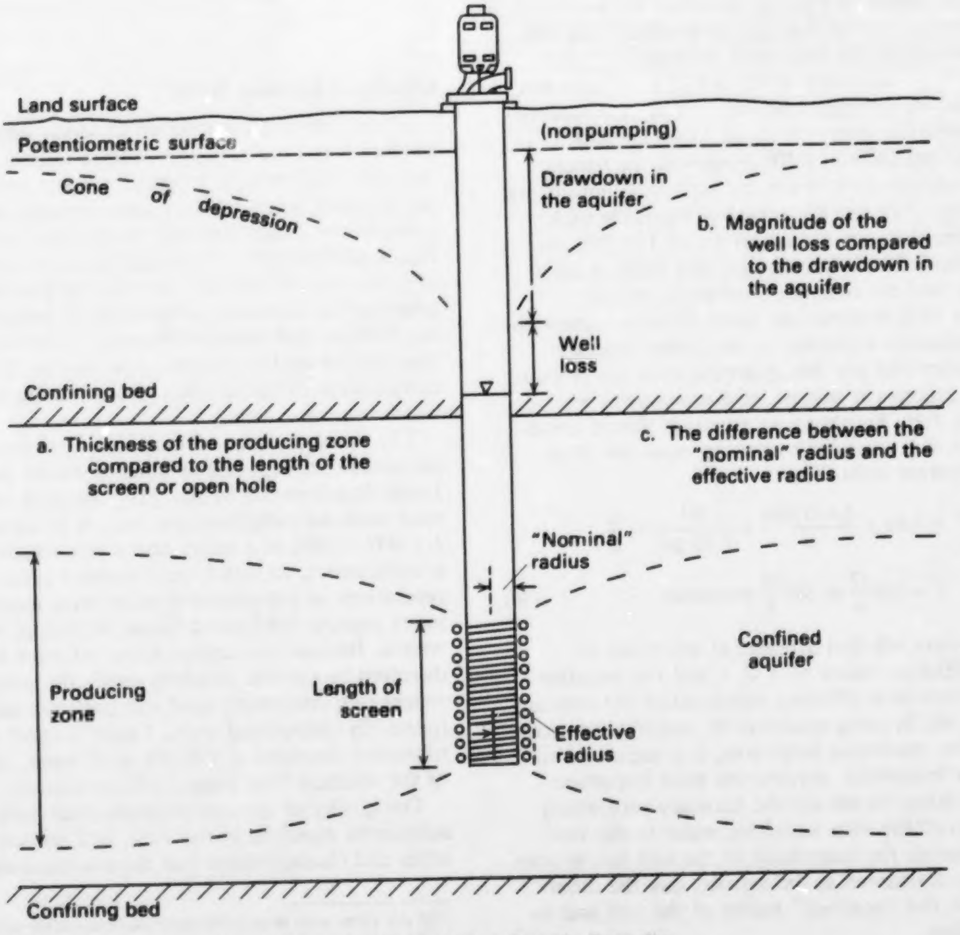

Land surface
Potentiometric surface
(nonpumping)
Cone of depression
Drawdown in the aquifer
b. Magnitude of the well loss compared to the drawdown in the aquifer
Well loss
Confining bed
a. Thickness of the producing zone compared to the length of the screen or open hole
c. The difference between the "nominal" radius and the effective radius
"Nominal" radius
Confined aquifer
Producing zone
Length of screen
Effective radius
Confining bed

Figure 61. Factors Affecting Estimates of Transmissivity Based on Specific Capacity

necessary to determine values for u and, using a table of values of u (or $1/u$) and $W(u)$, determine the corresponding values for $W(u)$.

Values of u are determined by substituting in equation 45 values of T, S, r, and t that are representative of conditions in the area. To illustrate, assume, in an area under investigation and for which a large number of values of specific capacity are available, that:

1. The principal aquifer is confined, and aquifer tests indicate that it has a storage coefficient of about 2×10^{-4} and a transmissivity of about 11,000 ft^2d^{-1}.
2. Most supply wells are 8 in. (20 cm) in diameter (radius, 0.33 ft).
3. Most values of specific capacity are based on 12-hour well-acceptance tests (t = 0.5 d).

Substituting these values in equation 45, we obtain

$$u = \frac{r^2S}{4Tt} = \frac{(0.33\ \text{ft})^2 \times (2 \times 10^{-4})}{4 \times (11{,}000\ \text{ft}^2\ \text{d}^{-1}) \times 0.5\ \text{d}}$$

$$u = \frac{2.22 \times 10^{-5}\ \text{ft}^2}{2.2 \times 10^4\ \text{ft}^2} = 1.01 \times 10^{-9}$$

A table of values of $W(u)$ for values of $1/u$ is contained in the section of this chapter entitled "Aquifer Tests." Therefore, the value of u determined above must be converted to $1/u$, which is 9.91×10^8, and this value is used to determine the value of $W(u)$. Values of $W(u)$ are given for values of $1/u$ of 7.69×10^8 and 10×10^8 but not for 9.91×10^8. However, the value of 10 is close enough to 9.91 for the purpose of estimating transmissivity from specific capacity. From the table, we determine that, for a value of $1/u$ of 10×10^8, the value of $W(u)$ is 20.15. Substituting this value in equation 44, we find the constant $W(u)/4\pi$ to be 1.60.

Equation 44 is in consistent units. However, transmissivity is commonly expressed in the United States in units of square feet per day, pumping rates are reported in units of gallons per minute, and drawdowns are measured in feet. To obtain an equation that is convenient to use, it is desirable to convert equation 44 to these inconsistent units. Thus

$$T = 1.60 \times \frac{1{,}440\ \text{min}}{\text{d}} \times \frac{\text{ft}^3}{7.48\ \text{gal}} \times \frac{Q}{s}$$

$$T = 308\frac{Q}{s}\ \text{or}\ 300\frac{Q}{s}\ \text{(rounded)} \qquad (46)$$

Many readers will find it useful at this point to substitute different values of T, S, r, and t in equation 45 to determine how different values affect the constant in equation 46. In using equation 46, modified as necessary to fit the conditions in an area, it is important to recognize its limitations. Among the most important factors that affect its use are the accuracy with which the thickness of the zone supplying water to the well can be estimated, the magnitude of the well loss in comparison with drawdown in the aquifer, and the difference between the "nominal" radius of the well and its effective radius.

Relative to these factors, the common practice is to assume that the value of transmissivity estimated from specific capacity applies only to the screened zone or to the open hole. To apply this value to the entire aquifer, the transmissivity is divided by the length of the screen or open hole (to determine the hydraulic conductivity per unit of length), and the result is multiplied by the entire thickness of the aquifer. The value of transmissivity determined by this method is too large if the zone supplying water to the well is thicker than the length of the screen or the open hole. Similarly, if the effective radius of the well is larger than the "nominal" radius (assuming that the "nominal" radius is used in equation 45), the transmissivity based on specific capacity again will be too large.

On the other hand, if a significant part of the drawdown in the pumping well is due to well loss, the transmissivity based on specific capacity will be too small. Whether the effect of all three of these factors cancels depends on the characteristics of both the aquifer and the well. Where a sufficient number of aquifer tests have been conducted, it may be feasible to utilize the results to modify the constant in equation 46 to account for the effect of these factors.

Quality of Ground Water

Water frequently is referred to as the universal solvent because it has the ability to dissolve at least small amounts of almost all substances that it contacts. Of the domestic water used by man, ground water usually contains the largest amounts of dissolved solids. As Figure 62 illustrates, the composition and concentration of substances dissolved in unpolluted ground water depend on the chemical composition of precipitation, on the biologic and chemical reactions occurring on the land surface and in the soil zone, and on the mineral composition of the aquifers and confining beds through which the water moves.

The concentrations of substances dissolved in water are commonly reported in units of weight per volume. In the International System (SI), the most commonly used units are milligrams per liter. A milligram equals 1/1,000 (0.001) of a gram, and a liter equals 1/1,000 of a cubic meter, so that 1 mg/l equals 1 gram m^{-3}.[2] Concentrations of substances in water were reported for many years in the United States in units of weight per weight. Because the concentration of most substances dissolved in water is relatively small, the weight per weight unit commonly used was parts per million (ppm). In inch-pound units, 1 ppm is equal to 1 lb of a substance dissolved in 999,999 lb of water, the weight of the solution thus being 1 million pounds.

The quality of ground water depends both on the substances dissolved in the water and on certain properties and characteristics that these substances impart to the water. Table 6 contains information on dissolved inorganic substances that normally occur in the largest concentrations and are most likely to affect water use. Table 7 lists other characteristics of water that are commonly reported in water analyses and that may affect water use.

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[2] To put these units in possibly more understandable terms, 1 mg/l equals 1 oz of a substance dissolved in 7,500 gal of water.

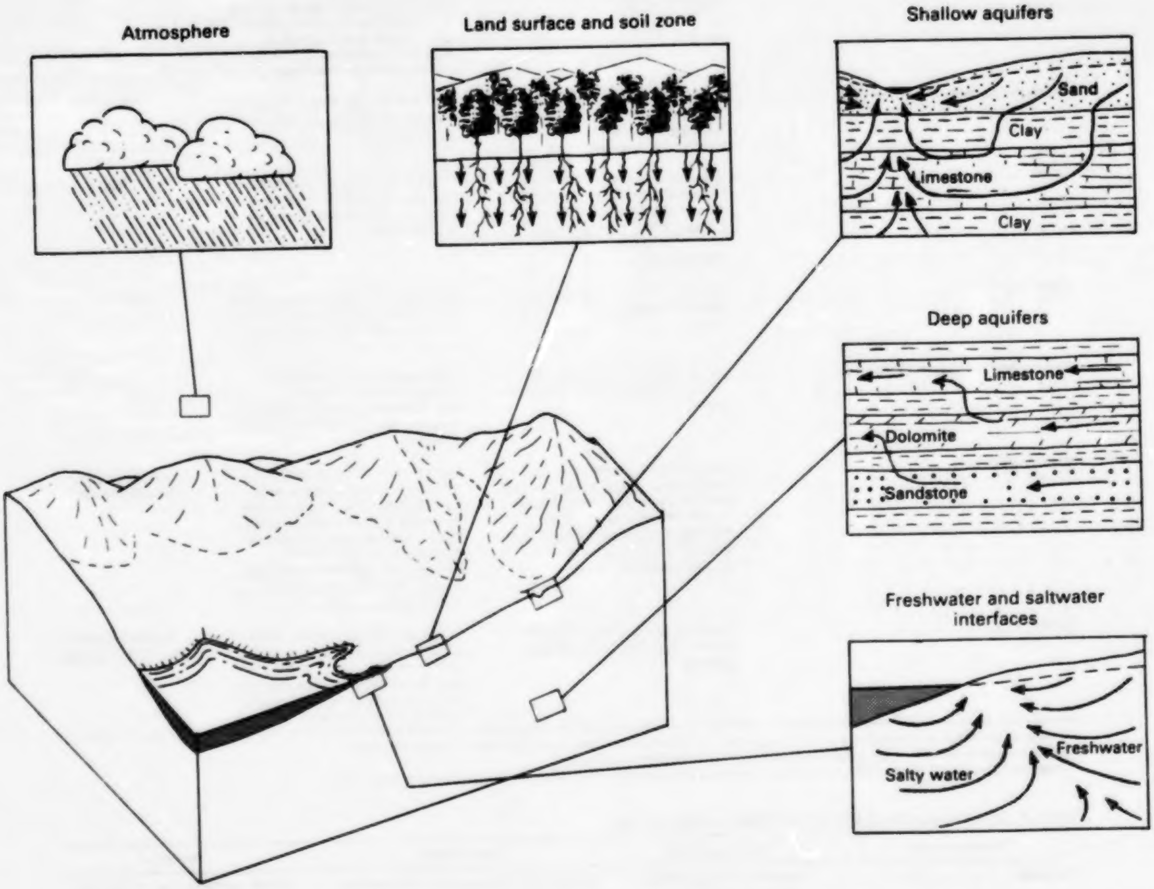


Figure 62. The Chemical Characteristics of Ground Water are Determined by the Chemical and Biological Reactions in the Zones Through Which the Water Moves

Table 6. Natural Inorganic Constituents Commonly Dissolved in Water That are Most Likely to Affect Use of the Water

Substance	Major natural sources	Effect on water use	Concentrations of significance (mg/l)[1]
Bicarbonate (HCO_3) and carbonate (CO_3)	Products of the solution of carbonate rocks, mainly limestone ($CaCO_3$) and dolomite ($CaMgCO_3$), by water containing carbon dioxide.	Control the capacity of water to neutralize strong acids. Bicarbonates of calcium and magnesium decompose in steam boilers and water heaters to form scale and release corrosive carbon dioxide gas. In combination with calcium and magnesium, cause carbonate hardness.	150–200
Calcium (Ca) and magnesium (Mg)	Soils and rocks containing limestone, dolomite, and gypsum ($CaSO_4$). Small amounts from igneous and metamorphic rocks.	Principal cause of hardness and of boiler scale and deposits in hot-water heaters.	25–50
Chloride (Cl)	In inland areas, primarily from seawater trapped in sediments at time of deposition; sition; in coastal areas, from seawater in contact with freshwater in productive aquifers.	In large amounts, increases corrosiveness of water and, in combination with sodium, gives water a salty taste.	250
Fluoride (F)	Both sedimentary and igneous rocks. Not widespread in occurrence.	In certain concentrations, reduces tooth decay; at higher concentrations, causes mottling of tooth enamel.	0.7–1.2[2]
Iron (Fe) and manganese (Mn)	Iron present in most soils and rocks; manganese less widely distributed.	Stain laundry and are objectionable in food processing, dyeing, bleaching, ice manufacturing, brewing, and certain other industrial processes.	Fe> 0.3, Mn> 0.05
Sodium (Na)	Same as for chloride. In some sedimentary rocks, a few hundred milligrams per liter may occur in freshwater as a result of exchange of dissolved calcium and magnesium for sodium in the aquifer materials.	See chloride. In large concentrations, may affect persons with cardiac difficulties, hypertension, and certain other medical conditions. Depending on the concentrations of calcium and magnesium also present in the water, sodium may be detrimental to certain irrigated crops.	69 (irrigation), 20–170 (health)[3]
Sulfate (SO_4)	Gypsum, pyrite (FeS), and other rocks containing sulfur (S) compounds.	In certain concentrations, gives water a bitter taste and, at higher concentrations, has a laxative effect. In combination with calcium, forms a hard calcium carbonate scale in steam boilers.	300–400 (taste), 600–1,000 (laxative)

[1] A range in concentration is intended to indicate the general level at which the effect on water use might become significant.

[2] Optimum range determined by the U.S. Public Health Service, depending on water intake.

[3] Lower concentration applies to drinking water for persons on a strict diet; higher concentration is for those on a moderate diet.

Table 7. Characteristics of Water that Affect Water Quality

Characteristic	Principal cause	Significance	Remarks
Hardness	Calcium and magnesium dissolved in the water.	Calcium and magnesium combine with soap to form an insoluble precipitate (curd) and thus hamper the formation of a lather. Hardness also affects the suitability of water for use in the textile and paper industries and certain others and in steam boilers and water heaters.	USGS classification of hardness (mg/l as $CaCO_3$): 0–60: Soft 61–120: Moderately hard 121–180: Hard More than 180: Very hard
pH (or hydrogen-ion activity)	Dissociation of water molecules and of acids and bases dissolved in water.	The pH of water is a measure of its reactive characteristics. Low values of pH, particularly below pH 4, indicate a corrosive water that will tend to dissolve metals and other substances that it contacts. High values of pH, particularly above pH 8.5, indicate an alkaline water that, on heating, will tend to form scale. The pH significantly affects the treatment and use of water.	pH values: less than 7, water is acidic; value of 7, water is neutral; more than 7, water is basic.
Specific electrical conductance	Substances that form ions when dissolved in water.	Most substances dissolved in water dissociate into ions that can conduct an electrical current. Consequently, specific electrical conductance is a valuable indicator of the amount of material dissolved in water. The larger the conductance, the more mineralized the water.	Conductance values indicate the electrical conductivity, in micromhos, of 1 cm^3 of water at a temperature of 25°C.
Total dissolved solids	Mineral substances dissolved in water.	Total dissolved solids is a measure of the total amount of minerals dissolved in water and is, therefore, a very useful parameter in the evaluation of water quality. Water containing less than 500 mg/l is preferred for domestic use and for many industrial processes.	USGS classification of water based on dissolved solids (mg/l): Less than 1,000: Fresh 1,000–3,000: Slightly saline 3,000–10,000: Moderately saline 10,000–35,000: Very saline More than 35,000: Briny

Saltwater Encroachment

In coastal areas, fresh ground water derived from precipitation on the land comes in contact with and discharges into the sea or into estuaries containing brackish water. The relation between the freshwater and the seawater, or brackish water, is controlled primarily by the differences in their densities.

The *density* of a substance is its mass per unit volume; thus, the density of water is affected by the amount of minerals, such as common salt (NaCl), that the water contains in solution. In metric units, the density of freshwater is about 1 gm cm^{-3}, and the density of seawater is about 1.025 gm cm^{-3}. Thus, freshwater, being less dense than seawater, tends to override or float on seawater.

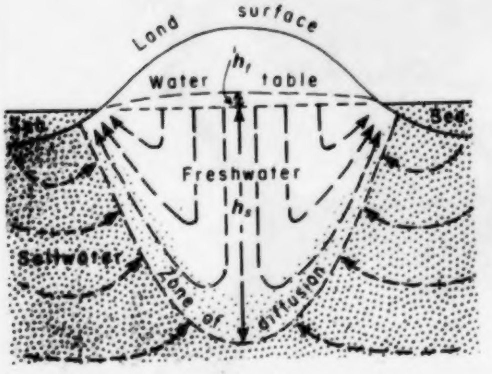


Figure 63. Freshwater Lens Floating on Saltwater

On islands, such as the Outer Banks of North Carolina, precipitation forms a freshwater lens that "floats" on the underlying saltwater, as illustrated in Figure 63. The higher the water table stands above sea level, the thicker the freshwater lens. This relation between the height of the water table and the thickness of the freshwater lens was discovered, independently, by a Dutchman, Badon Ghyben, and a German, B. Herzberg, and is referred to as the *Ghyben-Herzberg relationship.* This relation, expressed as an equation, is

$$h_s = \frac{\rho_f}{\rho_s - \rho_f}(h_f) \qquad (47)$$

where h_s is the depth of freshwater below sea level, ρ_f is the density of freshwater, ρ_s is the density of seawater, and h_f is the height of the water table above sea level.

On the basis of equation 47 and the differences between the densities of freshwater and seawater, the freshwater zone should extend to a depth below sea level (h_s) equal to 40 times the height of the water table above sea level (h_f). The Ghyben-Herzberg relation applies strictly, however, only to a homogenous and isotropic aquifer in which the freshwater is static and is in contact with a tideless sea or body of brackish water.

Figure 63 shows that tides cause saltwater to alternately invade and retreat from the freshwater zone, the result being a zone of diffusion across which the salinity changes from that of freshwater to that of seawater. A part of the seawater that invades the freshwater zone is entrained in the freshwater and is flushed back to the sea by the freshwater as it moves to the sea to discharge.

Because both the seawater and the freshwater are in motion (not static), the thickness of the freshwater zone in a homogenous and isotropic aquifer is greater than that predicted by the Ghyben-Herzberg equation. On the other hand, in a stratified aquifer (and nearly all aquifers are stratified), the thickness of the freshwater lens is less than that predicted because of the head loss incurred as the freshwater moves across the least permeable beds.

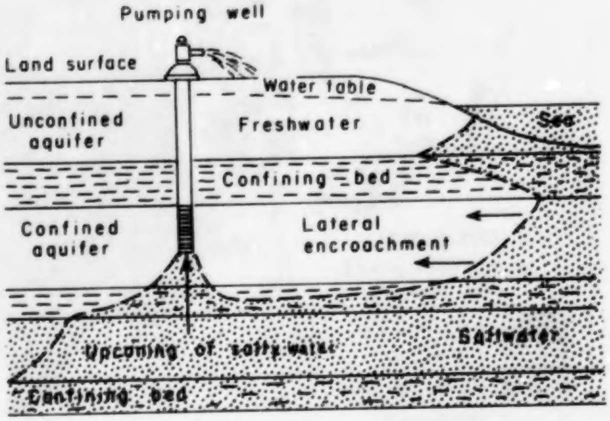
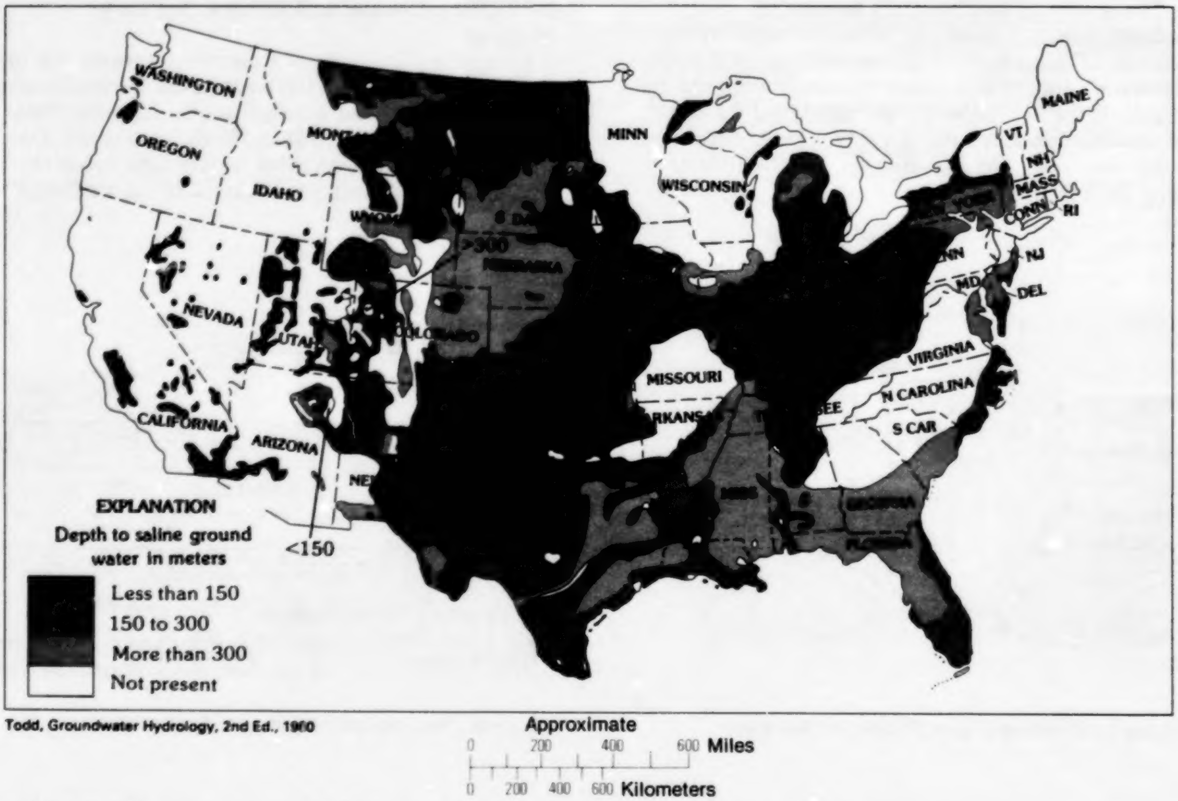


Figure 64. Two Aspects of Saltwater Encroachment

When freshwater heads are lowered by withdrawals through wells, the freshwater-saltwater contact migrates toward the point of withdrawals until a new balance is established, as shown in Figure 64. The movement of saltwater into zones previously occupied by freshwater is referred to as *saltwater encroachment.*

Saltwater encroachment is a serious problem in some coastal areas. *Upconing of salty water* beneath pumping wells is a more imminent problem than *lateral encroachment* in most areas. One reason is that lateral encroachment must displace a volume of freshwater much larger than that displaced by upconing. As Figure 65 shows, another reason is that approximately two-thirds of the United States is underlain by aquifers that yield water containing more than 1,000 mg/l of total dissolved solids. (See Table 7 in the "Quality of Ground Water" section of this chapter.) In most places, these aquifers are overlain by other aquifers that contain freshwater and that serve as sources of water supply. However, where supply wells are drilled too deeply or are pumped at too large a rate, upconing of the mineralized (salty) water may occur.

In the design of supply wells in areas underlain by or adjacent to salty water, consideration must be given to the possibility of saltwater encroachment. This consideration may involve selection of shallow aquifers or small pumping rates to avoid upconing or involve moving wells to more inland locations to avoid lateral encroachment.



Todd, Groundwater Hydrology, 2nd Ed., 1980

Figure 65. Depth to Ground Water Containing More Than 1000 mg/l of Total Dissolved Solids in the Conterminous United States

Temperature of Ground Water

The temperature of ground water is one of its most useful characteristics. Ground water has been used for many years on Long Island, N.Y., and at other places as a heat-exchange medium for air-conditioning systems. As a result of recent increases in energy costs, ground water is also now becoming increasingly important as a source of heat for "heat pumps."

The temperature of ground water responds to seasonal variations in the heat received at the Earth's surface from the Sun and by movement of heat from the Earth's interior. Figure 66 shows that the seasonal movement of heat into and out of the upper layers of the Earth's crust causes a seasonal fluctuation in ground-water temperatures to a depth of 10 to 25 m. The fluctuation is greatest near the surface, amounting to 5° to 10°C at depths of a few to several meters. In the zone affected by seasonal fluctuations, the mean annual ground-water temperature is 1° to 2°C higher than the mean annual air temperature. Consequently, a map showing the mean annual temperature of shallow ground water can be prepared on the basis of mean annual air temperature, as Figure 67 illustrates (based on a map showing mean annual air temperature prepared by the National Weather Service).

Figure 66 shows that movement of heat from the Earth's interior causes ground-water temperatures to increase with depth. This increase is referred to as the *geothermal gradient* and ranges from about 1.8°C per 100 m in areas underlain by thick sections of sedimentary rocks to about 3.6°C per 100 m in areas of recent volcanic activity. The effect of the geothermal

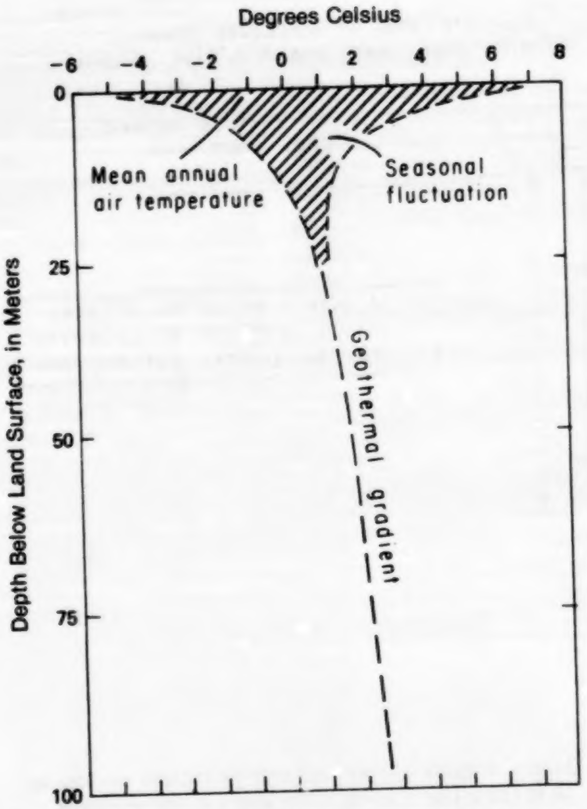


Figure 66. Changes in Ground-Water Temperature With Depth

gradient is not readily apparent in the zone affected by seasonal temperature fluctuations.

Movement of ground water causes a distortion in isotherms (lines depicting equal temperatures). This effect is most noticeable where ground-water withdrawal induces a movement of water from a stream into an aquifer. The distortion in ground-water temperature is most pronounced in the more permeable zones of the aquifer.

Protection of Supply Wells

Most, if not all, States have laws related to the location and construction of public-supply wells. Figure 68 shows typical requirements for supply wells. These laws and the rules and regulations developed for their administration and enforcement are concerned, among other things, with protecting supply wells from pollution. Pollution of the environment results from man's activities, and, consequently, except where deep wells or mines are used for waste disposal, it primarily affects the land surface, the soil zone, and the upper part of the saturated (ground water) zone. Therefore, the protection of supply wells includes avoiding areas that are presently polluted and sealing the wells in such a way as to prevent pollution in the future.

Fortunately, most ground-water pollution at the present time affects only relatively small areas that can be readily avoided in the selection of well sites. Among the areas in which at least shallow ground-water pollution should be expected are:

1. **Industrial districts** that include chemical, metal-

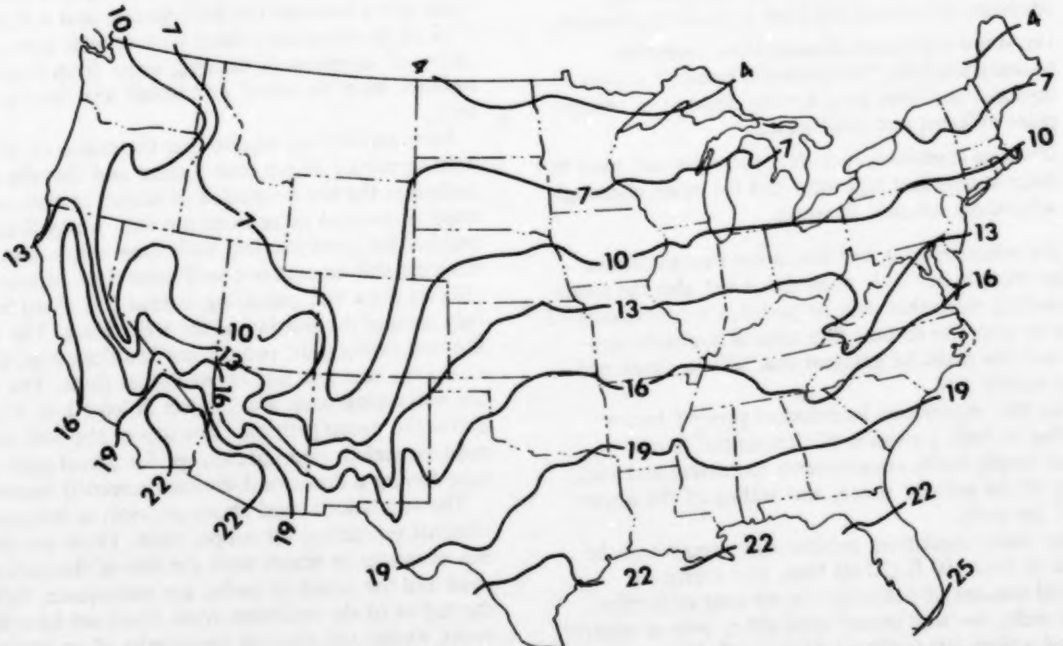


Figure 67. Approximate Temperature of Ground Water, in Degrees Celsius, in the Conterminous United States at Depths of 10 to 25 m

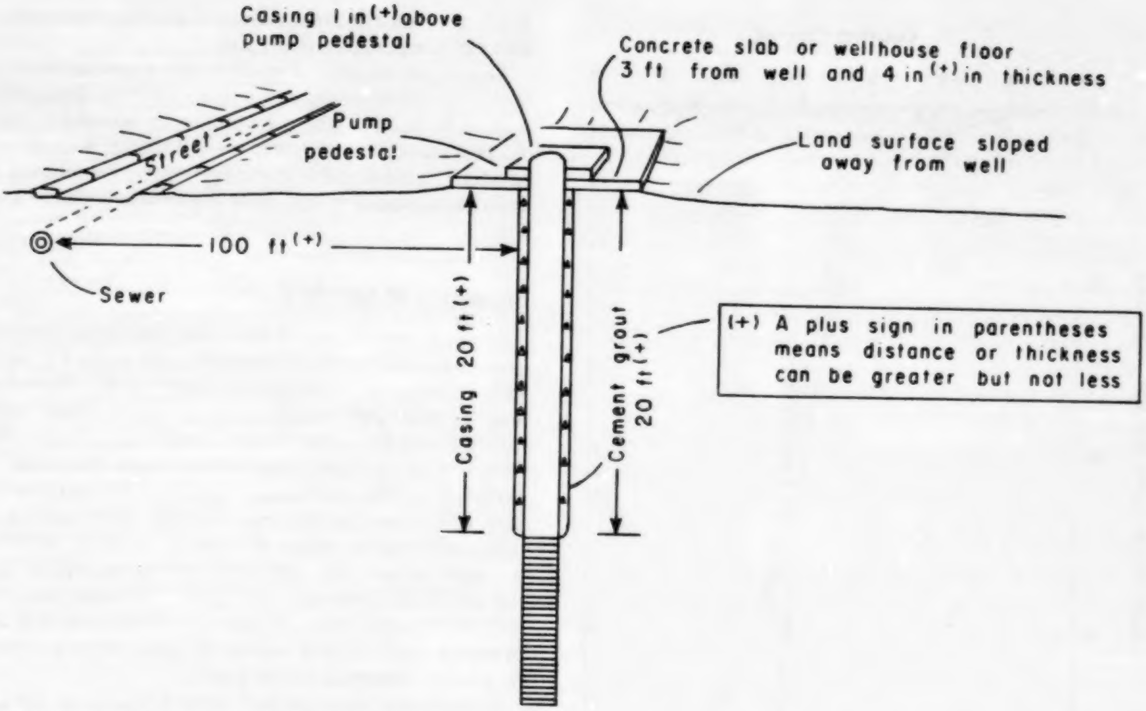

Casing 1 in(+) above pump pedestal
Concrete slab or wellhouse floor 3 ft from well and 4 in(+) in thickness
Street
Pump pedestal
Land surface sloped away from well
100 ft(+)
Sewer
Casing 20 ft(+)
Cement grout 20 ft(+)
(+) A plus sign in parentheses means distance or thickness can be greater but not less

Figure 68. Typical Requirements for Supply Wells

working, petroleum-refining, and other industries that involve fluids other than cooling water.

2. **Residential areas** in which domestic wastes are disposed of through septic tanks and cesspools.
3. **Animal feedlots** and other areas in which large numbers of animals are kept in close confinement.
4. **Liquid and solid waste disposal sites**, including sanitary landfills, "evaporation ponds," sewage lagoons, and sites used for the disposal of sewage-plant effluent and solid wastes.
5. **Chemical stockpiles**, including those for salt used to deice streets and highways and for other chemical substances soluable in water.

In the selection of a well site, areas that should be avoided include not only those listed but also the zones surrounding them that may be polluted by movement of wastes in response to both the natural hydraulic gradient and the artificial gradient that will be developed by the supply well.

Rules and regulations intended to prevent future pollution include provision of "exclusion" zones around supply wells, requirements for casing and for sealing of the annular space, and sealing of the upper end of the wells.

Many State regulations require that supply wells be located at least 100 ft (30 m) from any sources or potential sources of pollution. In the case of public-supply wells, the well owner must either own or control the land within 100 ft (30 m) of the well. In some States, a public-supply well may be located as close as 50 ft (15 m) to a sewer if the joints in the sewerline meet water-main standards.

Some State regulations require that all supply wells be cased to a depth of at least 20 ft (6 m) and that the annular space between the land surface and a depth of 20 ft (6 m) be completely filled with cement grout. The casing of supply wells drawing water from fractured bedrock must be seated and sealed into the top of the rock.

Most regulations require that the casing of all supply wells terminate above land surface and that the land surface at the site be graded or sloped so that surface water is diverted away from the well. Many States also require that public-supply wells have a continuous-bond concrete slab or concrete wellhouse floor at least 4 in. (10 cm) thick and extending at least 3 ft (1 m) horizontally around the outside of the well casing. The top of the well casing must project not less than 6 in. (15 cm) above the concrete slab or wellhouse floor. The top of the well casing must also project at least 1 in. (2.5 cm) above the pump pedestal. The top of the well casing must be sealed watertight except for a vent pipe or vent tube having a downward-diverted screened opening.

The regulations cited above provide, at best, only minimal protection for supply wells. There are numerous situations in which both the size of the exclusion zone and the depth of casing are inadequate. Relative to the radius of the exclusion zone, there are no arbitrary limits, except the physical boundaries of an aquifer, past which ground water cannot move. Relative to the minimum required casing, there are no vertical limits, except for the impermeable base of the ground-water system, past which polluted water cannot move.

On the other hand, there are geologic and hydrologic situations in which these regulations may be unnecessarily restrictive. An example is pollution in an unconfined aquifer down the hydraulic gradient from a supply well drawing from a deep confined aquifer overlain by a nonleaky confining bed.

Because of these factors, it is essential that officials involved in regulating the location and construction of supply wells be adequately trained in the fields of ground-water geology and hydrology so that they can protect the public health on the basis of scientific knowledge and technical judgment rather than on blind application of arbitrary regulations.

Well Records and Files

The collection and preservation of records on the construction, operation, maintenance, and abandonment of supply wells are an essential but largely neglected activity. This responsibility rests largely on the well owner or operator. The consequence of this neglect is that it is not possible to identify and to economically correct problems of declining yield or deterioration in water quality, and the design of new wells cannot incorporate past operational experience.

A file should be established on each supply well at the time when plans for its construction are initiated. From the initial planning to the final abandonment of the well, the following records should be generated and carefully preserved in this file:

1. **Initial design**, including drawings or written specifications on diameter, proposed total depth, position of screens or open hole, method of construction, and materials to be used in construction.
2. **Construction record**, including the method of construction and the driller's log and geophysical log of the materials penetrated during construction, the diameter of casings and screens, the slot size and metallic composition of screens, the depths of casing and screens, the total depth of the well, and the weight of the casing. Records and logs should also be retained for all test wells, including those that were not successful because of small yields.
3. **Well-acceptance test**, including a copy of the water-level measurements made before, during, and after the drawdown (pumping) test, a record of the pumping rate or rates, copies of any graphs of the data, and a copy of the hydrologist's report on the interpretation of the test results. (See the "Well-Acceptance Tests and Well Efficiency" section of this chapter.)
4. **Pump and installation data**, including the type of pump, the horsepower of the motor, the depth to the pump intake, a copy of the pump manufacturer's performance and efficiency data, and data on the length of the air line or a description of facilities provided for water-level measurements, including a description of the measuring point.
5. **Operating record**, including data on the type of meter used to measure the flow rate, weekly readings of the flowmeter dial, weekly measurements of the static and pumping water levels, and periodic analyses of water quality.
6. **Record of well maintenance**, including the dates and the activities instituted to increase the yield or to improve the water quality and data showing the results achieved.
7. **Record of well abandonment**, including the date that use of the well was discontinued and description of the methods and materials used to seal or plug the well.

The type of forms used for the records described above is not of critical importance. It is more important that the records be collected, regardless of the type of form that is used. It is important, however, that the date and the watch time be noted with each measurement of pumping rate and depth to water and on each water sample collected for water-quality analyses.

Numbers, Equations, and Conversions

The preceding discussions of basic ground-water hydrology involve the use of equations and physical units with which some readers may not be familiar. This discussion of numbers, equations,and conversion of units from one system of measurement to another is included for the benefit of those readers and for others who need to refresh their memories.

Expressing Large Numbers

$1{,}000 = 10 \times 10 \times 10 = 1 \times 10^3$

$1{,}000{,}000 = 10 \times 10 \times 10 \times 10 \times 10 \times 10 = 1 \times 10^6$

The numbers 3 and 6 are called exponents and indicate the number of times that 10 must be multiplied by itself to obtain the initial number.

Expressing Small Numbers

$$0.001 = \frac{1}{1{,}000} = \frac{1}{1 \times 10^3} = 1 \times 10^{-3}$$

$$0.000001 = \frac{1}{1{,}000{,}000} = \frac{1}{1 \times 10^6} = 1 \times 10^{-6}$$

Exponents in the denominator acquire a negative sign when they are moved to the numerator.

Simplifying Equations

Symbols in equations have numerical values and, in most cases, units of measurement, such as meters and feet, in which the values are expressed. For example, Darcy's law, one of the equations used in basic ground-water hydrology, is

$$Q = KA\left(\frac{dh}{dl}\right) \tag{6}$$

In metric units, hydraulic conductivity (K) is in meters per day, area (A) is in square meters, and hydraulic gradient (dh/dl) is in meters per meter. Substituting these units in Darcy's law, we obtain

$$Q = \frac{\text{meters}}{\text{day}} \times \text{meters}^2 \times \frac{\text{meters}}{\text{meters}} = \frac{\text{meters}^4}{\text{meters day}} = \text{m}^{4-1}\,\text{d}^{-1} = \text{m}^3\,\text{d}^{-1}$$

Similarly, in inch-pound units, K is in feet per day, A is in square feet, and dh/dl is in feet per feet. Substituting these units in Darcy's law, we obtain

$$Q = \frac{\text{feet}}{\text{day}} \times \text{feet}^2 \times \frac{\text{feet}}{\text{feet}} = \frac{\text{feet}^4}{\text{feet day}} = \text{ft}^{4-1}\,\text{d}^{-1} = \text{ft}^3\,\text{d}^{-1}$$

The characteristics of exponents are the same, whether they are used with numbers or with units of measurement. Exponents assigned to units of measurement are understood to apply, of course, to the value that the unit of measurement has in a specific problem.

Conversion of Units

Units of measurements used in ground-water literature are gradually changing from the inch-pound units of gallons, feet, and pounds to the International System of units of meters and kilograms (metric units). It is, therefore, increasingly important that those who use this literature become proficient in converting units of measurement from one system to another. Most conversions involve the fundamental principle that the numerator and denominator of a fraction can be multiplied by the same number (in essence, multiplying the fraction by 1) without changing the value of the fraction. For example, if both the numerator and the denominator of the fraction ¼ are multiplied by 2, the value of the fraction is not changed. Thus,

$$\frac{1}{4} \times \frac{2}{2} = \frac{2}{8} = \frac{1}{4} \text{ or } \frac{1}{4} \times \frac{2}{2} = \frac{1}{4} \times 1 = \frac{1}{4}$$

Similarly, to convert gallons per minute to other units of measurement, such as cubic feet per day, we must first identify fractions that contain both the units of time (minutes and days) and the units of volume (gallons and cubic feet) and that, when they are used as multipliers, do not change the numerical value. Relative to time, there are 1,440 minutes in a day. Therefore, if any number is multiplied by 1,440 min/d, the result will be in different units, but its numerical value will be unchanged. Relative to volume, there are 7.48 gallons in a cubic foot. Therefore, to convert gallons per minute to cubic feet per day, we multiply by these "unit" fractions, cancel the units of measurement that appear in both the numerator and the denominator, and gather together the units that remain. In other words, to convert gallons per minute to cubic feet per day, we have

$$\frac{\text{gallons}}{\text{minute}} = \frac{\text{gallons}}{\text{minute}} \times \frac{1{,}440\ \text{min}}{\text{d}} \times \frac{\text{cubic feet}}{7.48\ \text{gal}}$$

and, canceling gallons and minutes in the numerators and denominators, we obtain

$$\frac{\text{gallons}}{\text{minute}} = \frac{1{,}440\ \text{ft}^3}{7.48\ \text{d}} = 192.5\ \text{ft}^3\ \text{d}^{-1}$$

which tells us that 1 gal min^{-1} equals 192.5 $\text{ft}^3\ \text{d}^{-1}$.

We follow the same procedure in converting from inch-pound units to metric units. For example, to convert square feet per day to square meters per day, we proceed as follows:

$$\frac{\text{ft}^2}{\text{d}} = \frac{\text{ft}^2}{\text{d}} \times \frac{\text{m}^2}{10.76\ \text{ft}^2} = \frac{\text{m}^2}{10.76\ \text{d}} = 0.0929\ \text{m}^2\ \text{d}^{-1} = 9.29 \times 10^{-2}\ \text{m}^2\ \text{d}^{-1}$$

Table 8. Relation of Units of Hydraulic Conductivity, Transmissivity, Recharge Rates, and Flow Rates

Hydraulic Conductivity (K)

Meters per day (m d^{-1})	Centimeters per second (cm s^{-1})	Feet per day (ft d^{-1})	Gallons per day per square foot ($\text{gal d}^{-1}\ \text{ft}^{-2}$)
1	1.16×10^{-3}	3.28	2.45×10^{1}
8.64×10^{2}	1	2.83×10^{3}	2.12×10^{4}
3.05×10^{-1}	3.53×10^{-4}	1	7.48
4.1×10^{-2}	4.73×10^{-5}	1.34×10^{-1}	1

Transmissivity (T)

Square meters per day ($\text{m}^2\ \text{d}^{-1}$)	Square feet per day ($\text{ft}^2\ \text{d}^{-1}$)	Gallons per day per foot ($\text{gal d}^{-1}\ \text{ft}^{-1}$)
1	10.76	80.5
.0929	1	7.48
.0124	.134	1

Recharge Rates

Unit depth per year	Volume		
	($\text{m}^2\ \text{d}^{-1}\ \text{km}^{-2}$)	($\text{ft}^2\ \text{d}^{-1}\ \text{mi}^{-2}$)	($\text{gal d}^{-1}\ \text{mi}^{-2}$)
(In millimeters)	2.7	251	1,874
(In inches)	70	6,365	47,748

Flow Rates

($\text{m}^3\ \text{s}^{-1}$)	($\text{m}^3\ \text{min}^{-1}$)	($\text{ft}^3\ \text{s}^{-1}$)	($\text{ft}^3\ \text{min}^{-1}$)	(gal min^{-1})
1	60.	35.3	2,120	15,800
.0167	1	.588	35.3	264
.0283	1.70	1	60	449
.000472	.0283	.0167	1	7.48
.000063	.00379	.0023	.134	1

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A large number of publications on ground-water hydrology were consulted in the preparation of this report. A citation is shown in the text only where a publication was used as a specific source of tabular data.

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Chapter 3

Classification of Ground-Water Regions

To describe concisely ground-water conditions in the United States, it is necessary to divide the country into regions in which these conditions are generally similar. Because the presence and availability of ground water depends primarily on geologic conditions, ground-water regions also are areas in which the composition, arrangement, and structure of rock units are similar.[1]

To divide the country into ground-water regions, it is necessary to develop a classification that identifies features of ground-water systems that affect the occurrence and availability of ground water. The five features pertinent to such a classification are: (1) the components of the system and their arrangement, (2) the nature of the water-bearing openings of the dominant aquifer or aquifers with respect to whether they are of primary or secondary origin, (3) the mineral composition of the rock matrix of the dominant aquifers with respect to whether it is soluble or insoluble, (4) the water storage and transmission characteristics of the dominant aquifer or aquifers, and (5) the nature and location of recharge and discharge areas.

The first two of these features are primary criteria used in all delineations of ground-water regions. The remaining three are secondary criteria that are useful in subdividing what might otherwise be large and unwieldy regions into areas that are more homogenous and, therefore, more convenient for descriptive purposes. Table 9 lists each of the five features together with explanatory information. The fact that most of the features are more or less interrelated is readily apparent from the comments in the column headed "Significance of Feature".

Ground-Water Regions of the United States

On the basis of the criteria listed above the United States, exclusive of Alaska and Hawaii can be divided into 11 ground-water regions.

Figure 69 shows the boundaries of these 11 regions. A special area, region 12, which consists of those segments of the valleys of perennial streams that are underlain by sand and gravel thick enough to be hydrologically significant (thicknesses generally more than about 8 meters), is shown in Figure 70.

The nature and extent of the dominant aquifers and their relations to other units of the ground-water system are the primary criteria used in delineating the regions. Consequently, the boundaries of the regions generally coincide with major geologic boundaries and at most places do not coincide with drainage divides. Although this lack of coincidence emphasizes that the physical characteristics of ground-water systems and stream systems are controlled by different factors, it does not mean that the two systems are not related. Ground-water systems and stream systems are intimately related, as shown in the following discussions of each of the ground-water regions.

1. Western Mountain Ranges

(Mountains with thin soils over fractured rocks, alternating with narrow alluvial and, in part, glaciated valleys)

The Western Mountain Ranges, shown in Figure 71, encompass three areas totaling 708,000 km. The largest area extends in an arc from the Sierra Nevada in California, north through the Coast Ranges and Cascade Mountains in Oregon and Washington, and east and south through the Rocky Mountains in Idaho and Montana into the Bighorn Mountains in Wyoming and the Wasatch and Uinta Mountains in Utah. The second area includes the southern Rocky Mountains, which extend from the Laramie Range in southeastern Wyoming through central Colorado into the Sangre de Cristo Range in northern New Mexico. The smallest area includes the part of the Black Hills of South Dakota in which Precambrian rocks are exposed.

As would be expected in such a large region, both the origin of the mountains and the rocks that form them are complex. Most of the mountain ranges are underlain by granitic and metamorphic rocks flanked by consolidated sedimentary rocks of Paleozoic to Cenozoic age. The other ranges, including the San Juan Mountains in southwestern Colorado and the Cascade Mountains in Washington and Oregon, are underlain by lavas and other igneous rocks.

The summits and slopes of most of the mountains consist of bedrock exposures or of bedrock covered by a layer of boulders and other rock fragments produced by frost action and other weathering processes acting

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Table 9. Features of Ground-Water Systems Useful in the Delineation of Ground-Water Regions

Feature	Aspect	Range in Conditions	Significance of Feature
Component of the system	Unconfined aquifer	Thin, discontinuous, hydrologically insignificant. Minior aquifer, serves primarily as a storage reservoir and recharge conduit for underlying aquifer. The dominant aquifer.	Affect response of the system to pumpage and other stresses. Affect recharge and discharge conditions. Determine susceptibility to pollution.
	Confining beds	Not present, or hydrologically insignificant. Thin, markedly discontinuous, or very leaky. Thick, extensive, and impermeable. Complexly interbedded with aquifers or productive zones.	
	Confined aquifers	Not present, or hydrologically insignificant. Thin or not highly productive. Multiple thin aquifers interbedded with nonproductive zones. The dominant aquifer—thick and productive.	
	Presence and arrangements of components	A single, unconfined aquifer. Two interconnected aquifers of essentially equal hydrologic importance. A three-unit system consisting of an unconfined aquifer, a confining bed, and confined aquifer. A complexly interbedded sequence of aquifers and confining beds.	
Water-bearing openings of dominant aquifer	Primary openings	Pores in unconsolidated deposits. Pores in semiconsolidated rocks. Pores, tubes, and cooling fractures in volcanic (extrusive-igneous) rocks.	Control water-storage and transmission characteristics. Affect disperson and dilution of wastes.
	Secondary openings	Fractures and faults in crystalline and consolidated sedimentary rocks. Solution-enlarged openings in limestones and other soluble rocks.	
Composition of rock matrix of dominant aquifer	Insoluble	Essentially insoluble. Both relatively insoluble and soluble constituents.	Affects water-storage and transmission characteristics. Has major influence on water quality.
	Soluble	Relatively soluble.	
Storage and transmission characteristics of dominant aquifer	Porosity	Large, as in well-sorted, unconsolidated deposits. Moderate, as in poorly-sorted unconsolidated deposits and semiconsolidated rocks. Small, as in fractured crystalline and consolidated sedimentary rocks.	Control response to pumpage and other stresses. Determine yield of wells. Affect long-term yield of system. Affect rate at which pollutants move.
	Transmissivity	Large, as in cavernous limestones, some lava flows, and clean gravels. Moderate, as in well-sorted, coarse-grained sands, and semiconsolidated limestones. Small, as in poorly-sorted, fine-grained deposits and most fractured rocks. Very small, as in confining beds.	
Recharge and discharge conditions of dominant aquifer	Recharge	In upland areas between streams, particularly in humid regions. Through channels of losing streams. Largely or entirely by leakage across confining beds from adjacent aquifers.	Affect response to stress and long-term yields. Determine susceptibility to pollution. Affect water quality.
	Discharge	Through springs or by seepage to stream channels, estuaries, or the ocean. By evaporation on flood plains and in basin "sinks." By seepage across confining beds into adjacent aquifers.	

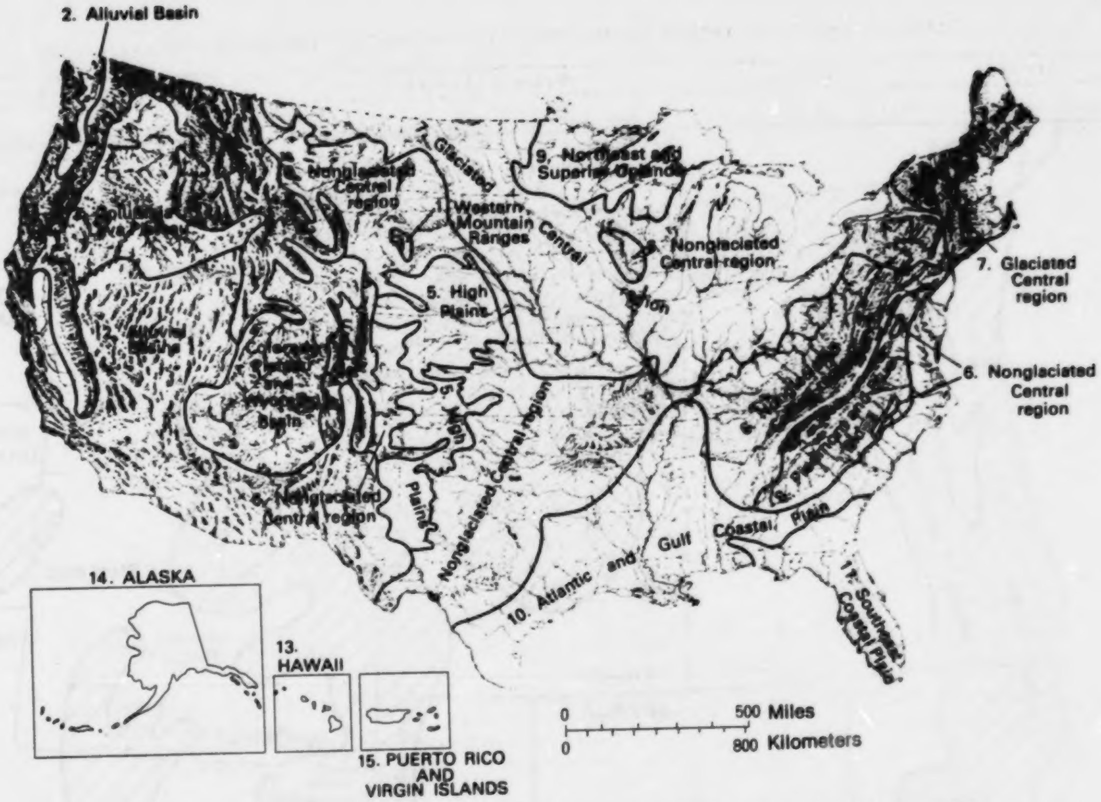
2. Alluvial Basin
Glaciated Central region
9. Northeast and Superior Uplands
8. Nonglaciated Central region
1. Western Mountain Ranges
8. Nonglaciated Central region
7. Glaciated Central region
5. High Plains
5. High Plains
6. Nonglaciated Central region
8. Nonglaciated Central region
Nonglaciated Central region
10. Atlantic and Gulf Coastal Plain
11. Southeast Coastal Plain
14. ALASKA
13. HAWAII
15. PUERTO RICO AND VIRGIN ISLANDS
0
500 Miles
0
800 Kilometers

Figure 69. Ground-Water Regions Used in This Report [The Alluvial Valleys region (region 12) is shown on figure 70]

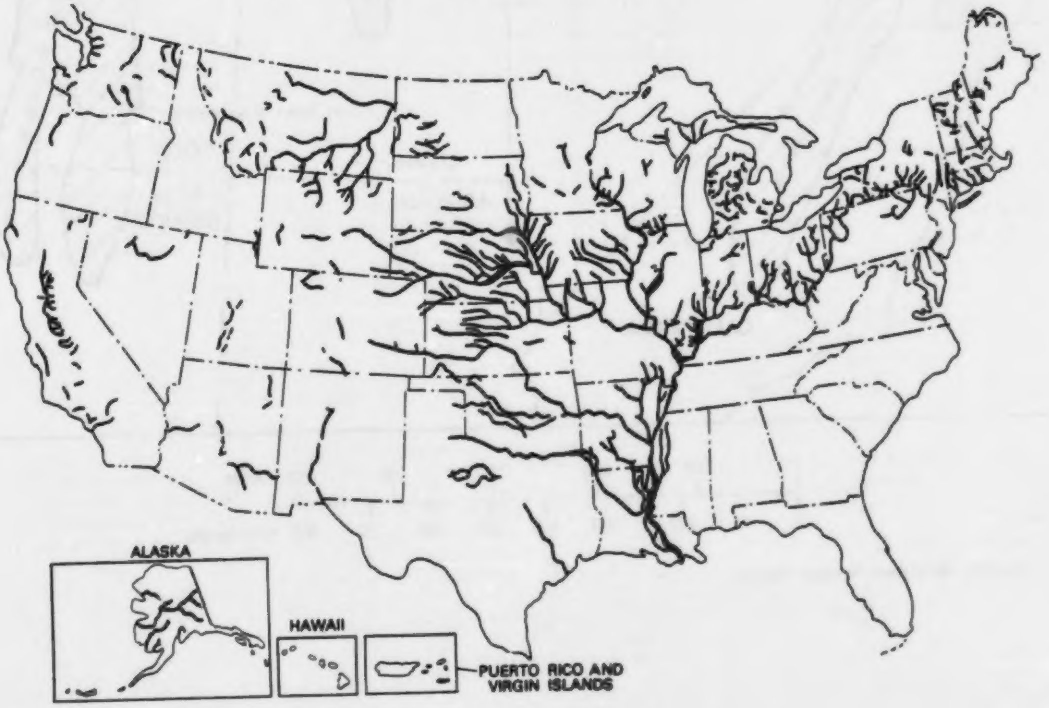

Figure 70. Alluvial Valleys Ground-Water Region

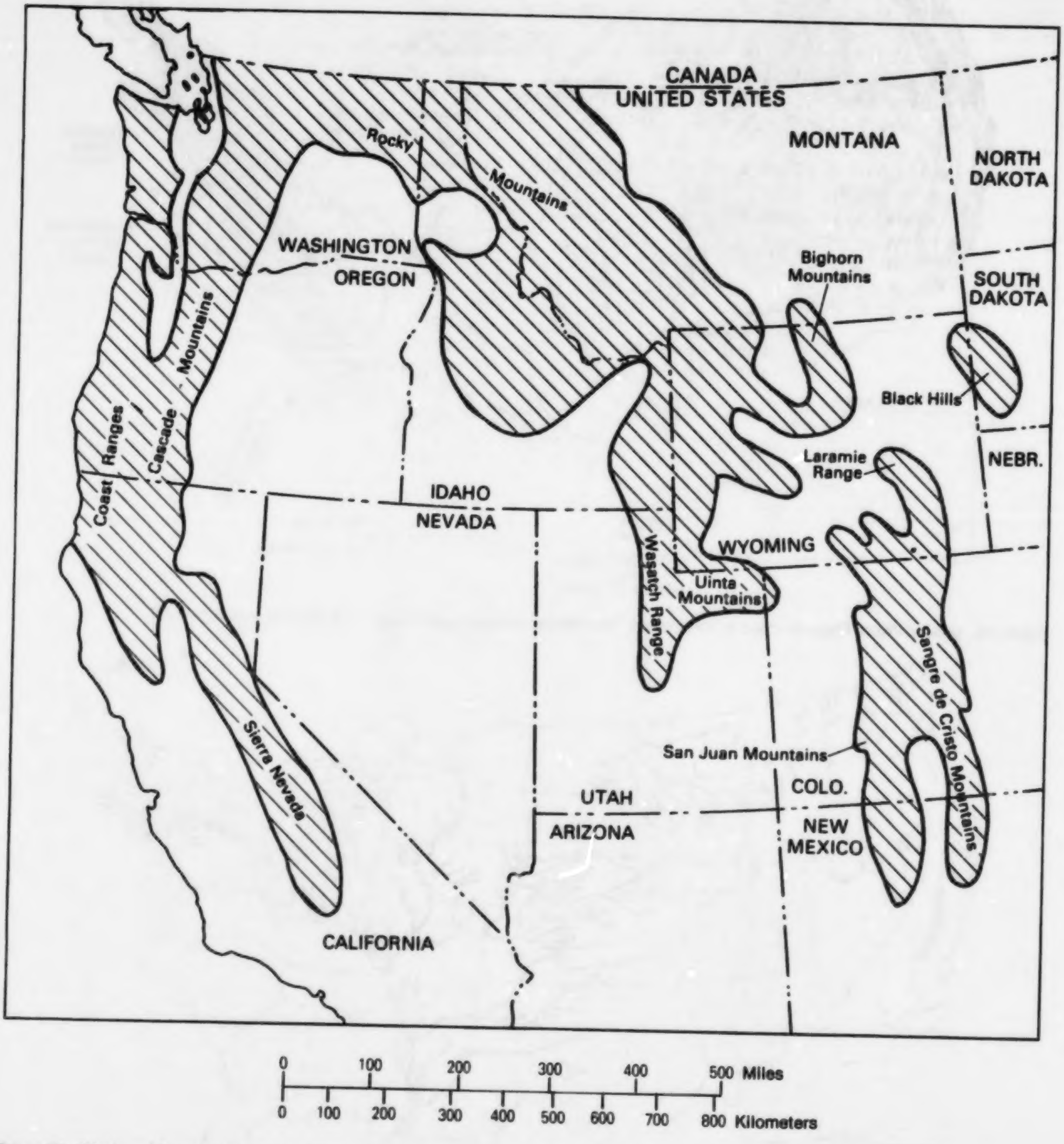

Figure 71. Western Mountain Ranges Region

on the bedrock. This layer is generally only a few meters thick on the upper slopes but forms a relatively thick apron along the base of the mountains. The narrow valleys are underlain by relatively thin, coarse, bouldery alluvium washed from the higher slopes. The large synclinal valleys and those that occupy downfaulted structural throughs are underlain by moderately thick deposits of coarse-grained alluvium transported by streams from the adjacent mountains, as shown in Figure 72.

The Western Mountain Ranges and the mountain ranges in adjacent regions are the principal sources of water supplies developed at lower altitudes in the western half of the conterninous United States. As McGuinness[2] noted, the mountains of the West are moist "islands" in a sea of desert or semidesert that covers the western half of the Nation. The heaviest precipitation falls on the western slopes; thus, these slopes are the major source of runoff and are also the most densely vegetated. Much of the precipitation falls as snow during the winter.

The Western Mountain Ranges are sparsely populated and have relatively small water needs. The region is an exporter of water to adjacent "have-not" areas. Numerous surface reservoirs have been constructed in the region. Many such impoundments have been developed on streams that drain the western flank of the Sierra Nevada in California and the Rocky Mountains in Colorado.

Melting snow and rainfall at the higher altitudes in the region provide abundant water for ground-water recharge. However, the thin soils and bedrock fractures in areas underlain by crystalline rocks fill quickly, and the remaining water runs off overland to streams. Because of their small storage capacity, the underground openings provide limited base runoff to the streams, which at the higher altitudes flow only during rains or snowmelt periods. Thus, at the higher altitudes in this region underlain by crystalline rocks, relatively little opportunity exists for development of ground-water supplies. The best opportunities exist in the valleys that contain at least moderate thicknesses of saturated alluvium or in areas underlain by permeable sedimentary or volcanic rocks. Ground-water supplies in the valleys are obtained both from wells drawing from the alluvium and from wells drawing from the underlying rocks. The yields of wells in crystalline bedrock and from small, thin deposits of alluvium are generally adequate only for domestic and stock needs. Large yields can be obtained from the alluvial deposits that overlie the major lowlands and from wells completed in permeable sedimentary or volcanic rocks.

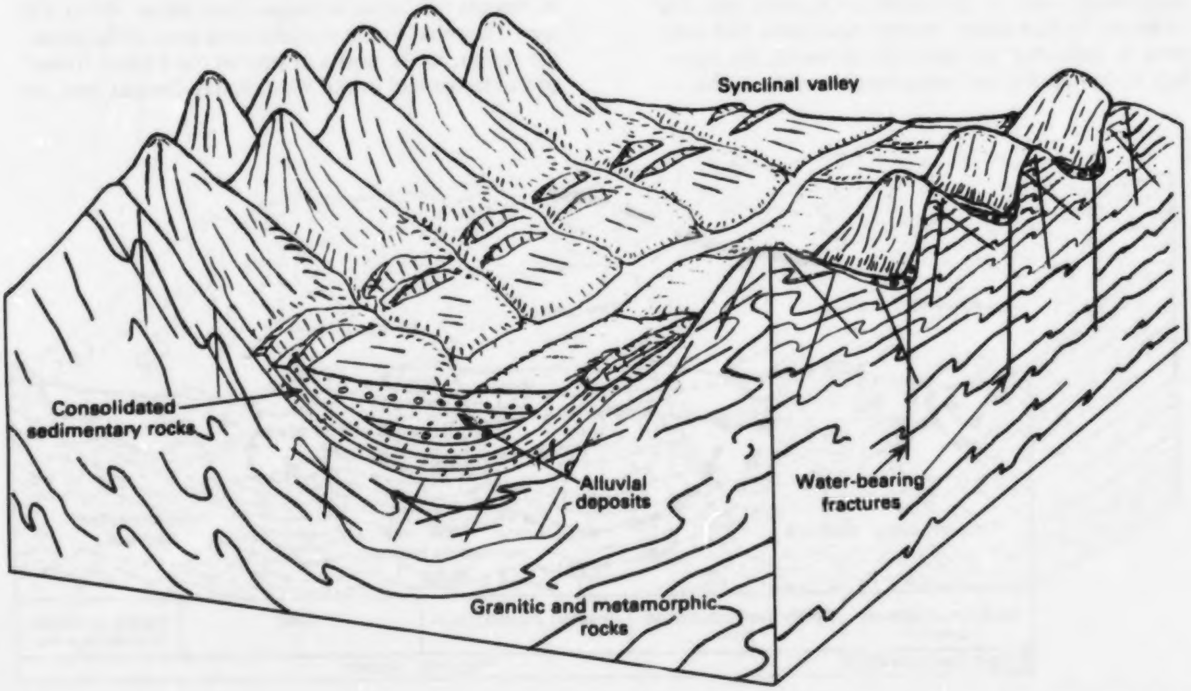
Synclinal valley
Consolidated sedimentary rocks
Alluvial deposits
Water-bearing fractures
Granitic and metamorphic rocks

Figure 72. Topographic and Geologic Features in the Southern Rocky Mountains Part of the Western Mountain Ranges Region

2. Alluvial Basins

(Thick alluvial deposits in basins and valleys bordered by mountains and locally of glacial origin)

The Alluvial Basins region occupies a discontinuous area of 1,025,000 km^2 extending from the Puget Sound-Williamette Valley area of Washington and Oregon to west Texas. The region consists of an irregular alternation of basins or valleys and mountain ranges. In the Alluvial Basins region, basins and valleys are the dominant feature. The principal exception is the Coast Ranges of southern California which topographically more closely resemble the Western Mountain Ranges.

Most of the Nevada and all of the Utah parts of this region are an area of internal drainage referred to as the Great Basin. No surface or subsurface flow leaves this part of the region and all water reaching it from adjacent areas and from precipitation is evaporated or transpired.

The basins and valleys range from about 85 m below sea level in Death Valley in California to 2,000 m above sea level in the San Luis Valley in Colorado. The basins range in size from a few hundred meters in width and a kilometer or two in length to, for the Central Valley of California, as much as 80 km in width and 650 km in length. The crests of the mountains are commonly 1,000 to 1,500 m above the adjacent valley floors.

The surrounding mountains, and the bedrock beneath the basins, consist of granite and metamorphic rocks of Precambrian to Tertiary age and consolidated sedimentary rocks of Paleozoic to Cenozoic age. The rocks are broken along fractures and faults that may serve as water-bearing openings. However, the openings in the granitic and metamorphic rocks in the mountainous areas have a relatively small capacity to store and to transmit ground water.

The dominant element in the hydrology of the region is the thick (several hundred to several thousand meters) layer of generally unconsolidated alluvial material that partially fills the basins. Figures 73, 74, and 75 illustrate this dominant element. Generally, the coarsest material occurs adjacent to the mountains; the material gets progressively finer toward the centers of the basins. However, as Figure 74 shows, in most alluvial fans there are layers of sand and gravel that extend into the central parts of the basins. In time, the fans formed by adjacent streams coalesced to form a continuous and thick deposit of alluvium that slopes gently from the mountains toward the center of the basins. These alluvial-fan deposits are overlain by or grade into fine-grained flood plain, lake, or playa deposits in the central part of most basins. The fine-grained deposits are especially suited to large-scale cultivation.

The Puget Sound and Williamette Valley areas differ geologically from the remainder of the region. The Puget Sound area is underlain by thick and very permeable deposits of gravel and sand laid down by glacial meltwater. The gravel and sand are interbedded with clay in parts of the area. The Williamette Valley is mostly underlain by interbedded sand, silt, and clay deposited on floodplains by the Williamette River and other streams.

The Alluvial Basins region is the driest area in the United States, with large parts of it being classified as semiarid and arid. Annual precipitation in the valleys in Nevada and Arizona ranges from about 100 to 400 mm. However, in the mountainous areas throughout the region, in the northern part of the Central Valley of California and in the Washington–Oregon area, an-

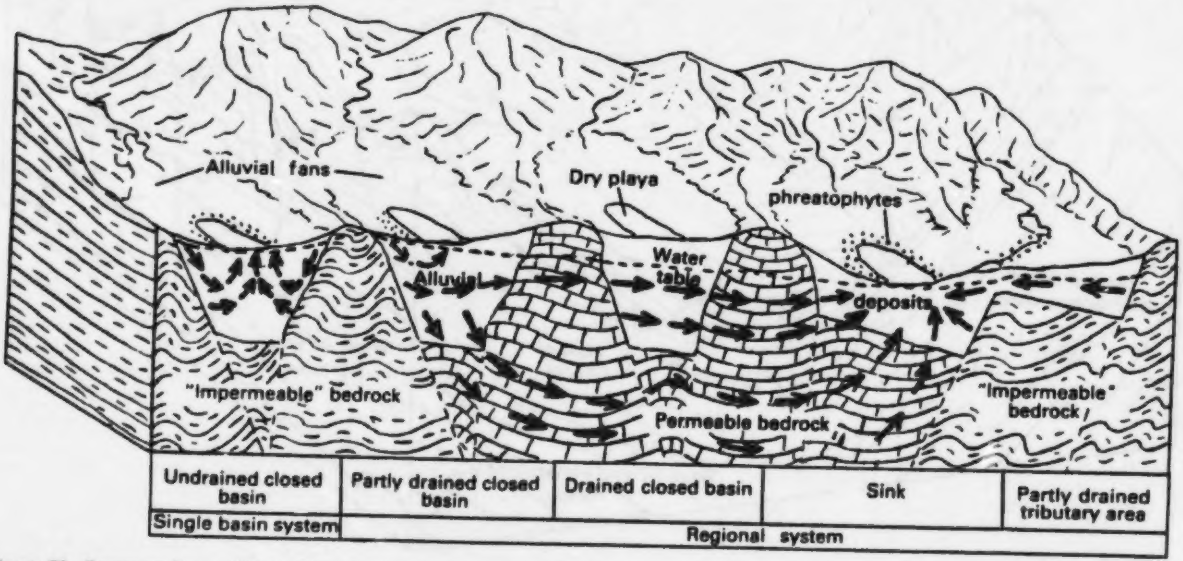

Figure 73. Common Ground-Water Flow Systems in the Alluvial Basins Region (From U.S. Geological Survey Professional Paper 813-G)[3]

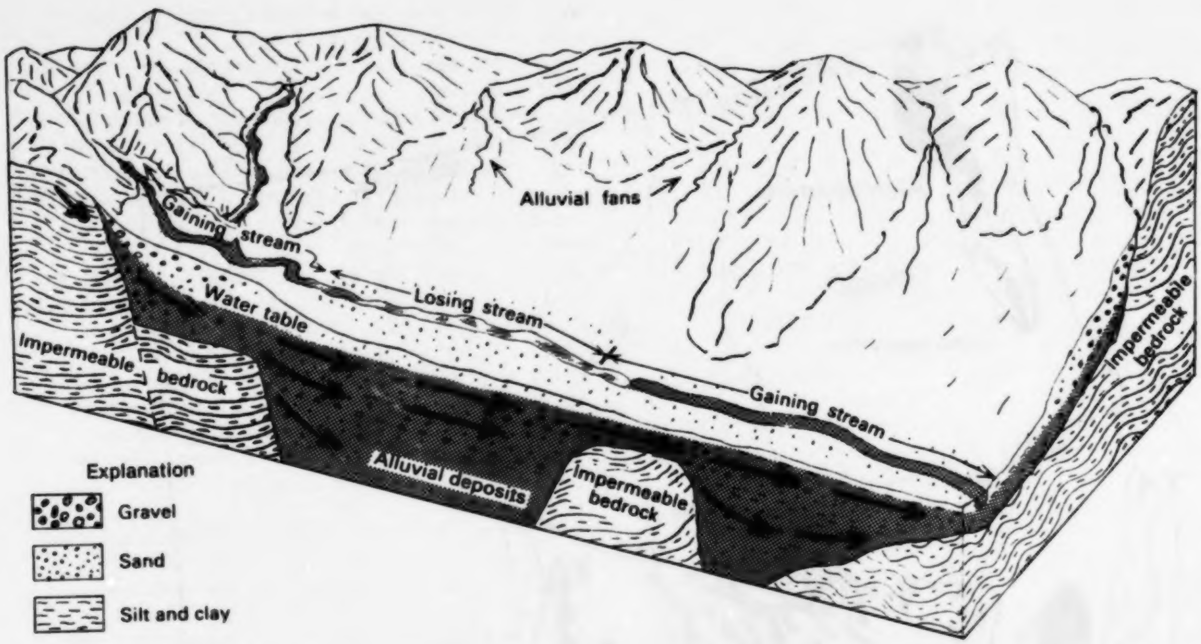

Figure 74. Common Relationships Between Ground Water and Surface Water in the Alluvial Basins Region (Modified from U.S. Geological Survey Professional Paper 813-G.)[3]

nual precipitation ranges from about 400 mm to more than 800 mm. The region also receives runoff from streams that originate in the mountains of the Western Mountain Ranges region.

Because of the very thin cover of unconsolidated material on the mountains, precipitation runs off rapidly down the valleys and out onto the fans, where it infiltrates. The water moves through the sand and gravel layers toward the centers of the basins. The centers of many basins consist of flat-floored, vegetation-free areas onto which ground water may discharge and on which overland runoff may collect during intense storms. The water that collects in these areas (playas), evaporates relatively quickly, leaving both a thin deposit of clay and other sediment and a crust of the soluble salts that were dissolved in the water, as Figure 73 illustrates.

Studies in the region have shown that the hydrology of the alluvial basins is more complex than that described in the preceding paragraph, which applies only to what has been described as "undrained closed basins". As Figure 73 shows, water may move through permeable bedrock from one basin to another, arriving, ultimately, at a large playa referred to as a "sink". Water discharges from sinks not by "sinking" into the ground, but by evaporating. In those parts of the region drained by perennial watercourses groundwater discharges to the streams from the alluvial deposits. However, before entering the streams, water may move down some valleys through the alluvial deposits for tens of kilometers. A reversal of this situation occurs along the lower Colorado River and at the upstream end of the valleys of some of the other perennial streams; in these areas, water moves from the streams into the alluvium to supply the needs of the adjacent vegetated zones.

Ground water is the major source of water in the Alluvial Basins region. Because of the dry climate, agriculture requires intensive irrigation. Most of the ground water is obtained from the sand and gravel deposits in the valley alluvium. These deposits are interbedded with finer grained layers of silt and clay that are also saturated with water. When hydraulic heads in the sand and gravel layers are lowered by withdrawals, the water in the silt and clay begins to move slowly into the sand and gravel. The movement, which in some areas takes decades to become significant, is accompanied by compaction of the silt and clay and subsidence of the land surface. Subsidence is most severe in parts of the Central Valley, where it exceeds 9 m in one area, and in southern Arizona, where subsidence of more than 4 m has been observed.

3. Columbia Lava Plateau

(Thick sequence of lava flows irregularly interbedded with thin unconsolidated deposits and overlain by thin soils)

As Figure 76 shows, the Columbia Lava Plateau occupies an area of 366,000 km^2 in northeastern California, eastern Washington and Oregon, southern Idaho, and northern Nevada. As its name implies, it is basically a plateau, standing generally between 500 and 1,800 m above sea level, that is underlain by a great

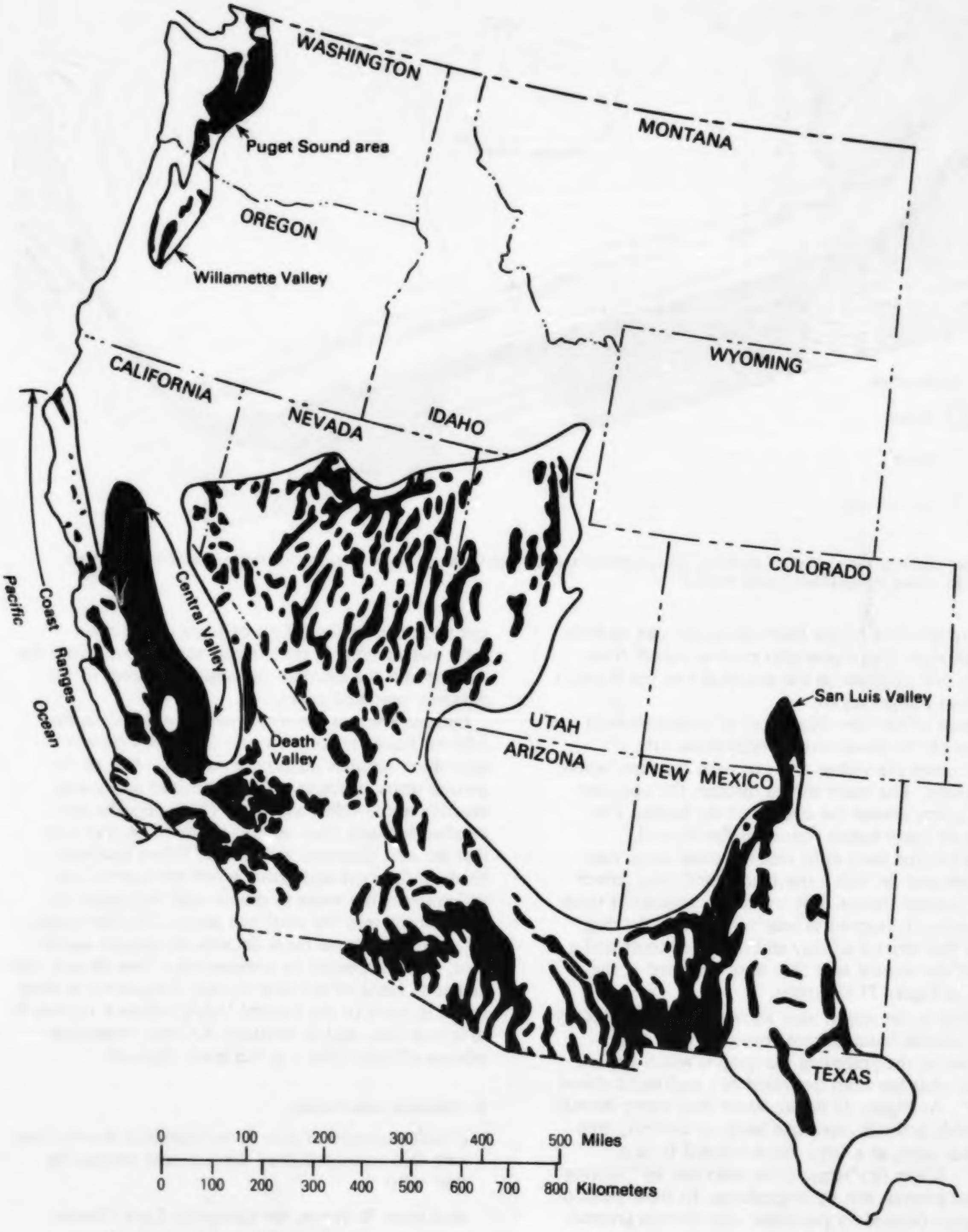

Figure 75. Areas Underlain by Sand and Gravel in the Alluvial Basins Region

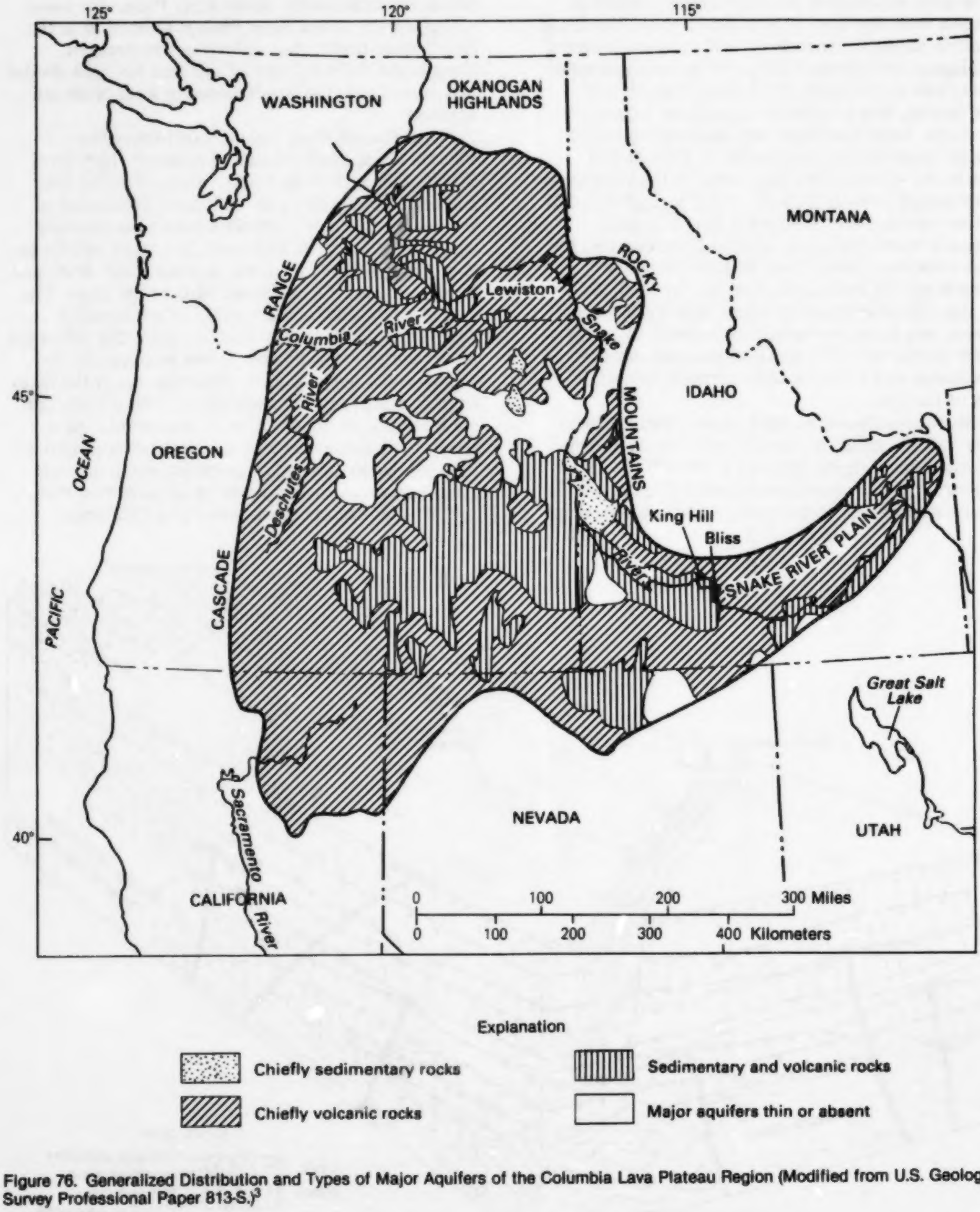

Figure 76. Generalized Distribution and Types of Major Aquifers of the Columbia Lava Plateau Region (Modified from U.S. Geological Survey Professional Paper 813-S.)[3]

thickness of lava flows irregularly interbedded with silt, sand, and other unconsolidated deposits.

The great sequence of lava flows, which ranges in thickness from less than 50 m adjacent to the bordering mountain ranges to more than 1,000 m in south-central Washington and southern Idaho, is the principal water-bearing unit in the region. As Figure 77 shows, the water-bearing lava is underlain by granite, metamorphic rocks, older lava flows, and sedimentary rocks, none of which are very permeable. Individual lava flows in the water-bearing zone range in thickness from several meters to more than 50 m and average about 15 m. The volcanic rocks yield water mainly from permeable zones that occur at or near the contacts between some flow layers. Parts of some flows are separated by soil zones and, at places, by sand, silt, and clay. These sedimentary layers, where they occur between lava flows, are commonly referred to as "interflow sediments". Gravel, sand, silt, and clay cover the volcanic rocks and the older exposed bedrock in parts of the area.

From the standpoint of the hydraulic characteristics of the volcanic rocks, it is useful to divide the Columbia Lava Plateau region into two parts: (1) the area in southeastern Washington, northeastern Oregon, and the Lewiston area of Idaho, part of which is underlain by volcanic rocks of the Columbia River Group; and (2) the remainder of the area shown on Figure 76, which also includes the Snake River Plain. The basalt underlying the Snake River Plain is referred to as the Snake River Basalt; that underlying southeastern Oregon and the remainder of this area has been divided into several units, to which names of local origin are applied.[4]

The Columbia River Group is of Miocene to Pliocene (?) age and consists of relatively thick flows that have been deformed into a series of broad folds and offset locally along normal faults. Movement of ground water occurs primarily through the interflow zones near the top of flows and, to a much smaller extent, through fault zones and through joints developed in the dense central and lower parts of the flows. The axes of sharp folds and the offset of the interflow zones along faults form subsurface dams that affect the movement of ground water. Water reaching the interflow zones tends to move down the dip of the flows from fold axes and to collect updip behind faults that are transverse to the direction of movement.[5] As a result, the basalt in parts of the area is divided into a series of barrier-controlled reservoirs, which are only poorly connected hydraulically to adjacent reservoirs.

The water-bearing basalt underlying California,

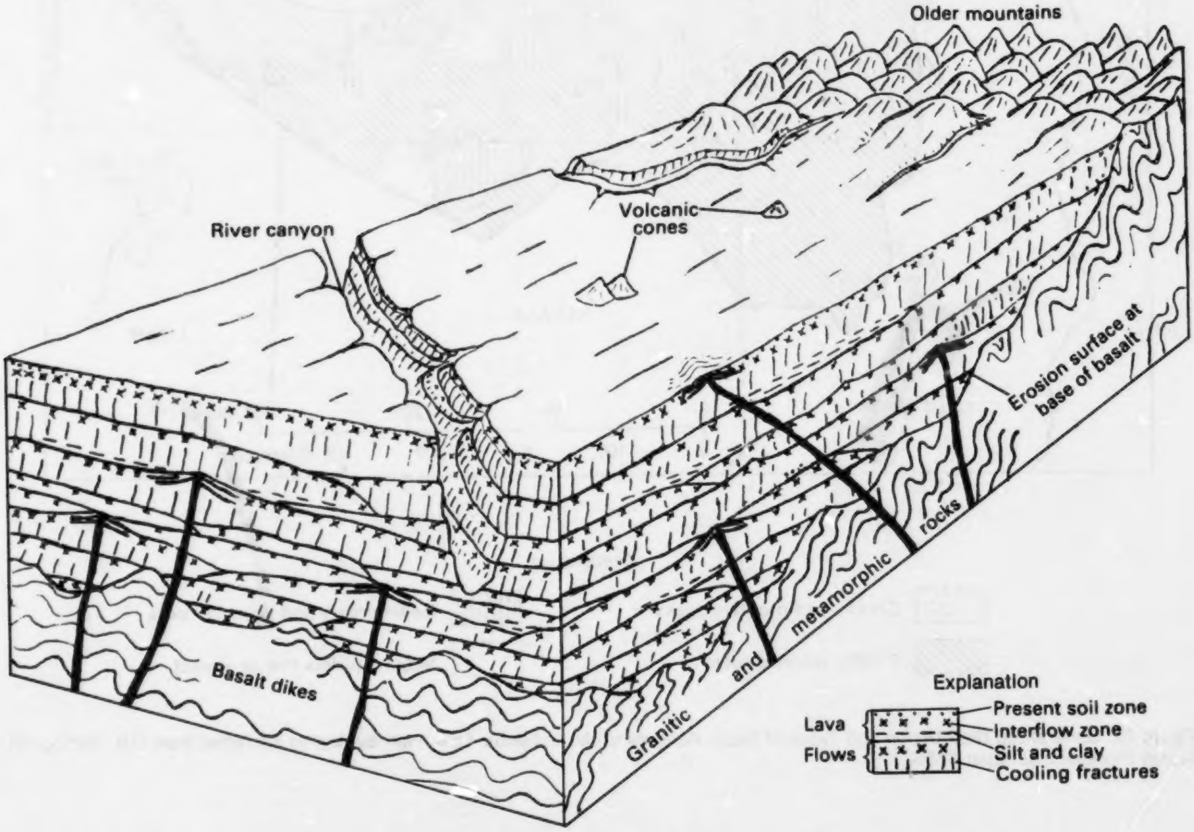

Figure 77. Topographic and Geologic Features of the Columbia Lava Plateau Region

Nevada, southeastern Oregon, and southern Idaho is of Pliocene to Holocene age and consists of small, relatively thin flows that have been affected to a much smaller extent by folding and faulting than has the Columbia River Group. The thin flows contain extensive, highly permeable interflow zones that are relatively effectively interconnected through a dense network of cooling fractures. Structural barriers to ground-water movement are of minor importance. This is demonstrated by conditions in the 44,000-square-kilometer area of the Snake River Plain east of Bliss, Idaho.

The interflow zones form a complex sequence of relatively horizontal aquifers that are separated vertically by the dense central and lower parts of the lava flows and by interlayered clay and silt. Hydrologists estimate that the interflow zones, which range in thickness from about 1 m to about 8 m, account for about 10 percent of the basalt. MacNish and Barker[6] have estimated that the hydraulic conductivity along the flow-contact zones may be a billion times larger than the hydraulic conductivity across the dense zones. The lateral extent of individual aquifers is highly variable.

The large differences in hydraulic conductivity between the aquifers and the intervening "confining zones" result in significant differences in hydraulic heads between different aquifers. These differences reflect the head losses that occur as water moves vertically through the system. As a result, heads decrease with increasing depth in recharge areas and increase with increasing depth near the streams that serve as major lines of ground-water discharge. As Figure 78 shows, the difference in heads between different aquifers can result in the movement of large volumes of water between aquifers through the openhold (uncased) sections of wells.

Much of the Columbia Lava Plateau region is in the "rain shadow" east of the Cascades and, as a result, receives only 200 to 1,200 mm of precipitation annually. The areas that receive the least precipitation include the plateau area immediately east of the Cascades and the Snake River Plain. Recharge to the ground-water system depends on several factors, including the amount and seasonal distribution of precipitation and the permeability of the surficial materials. Most precipitation occurs in the winter and thus coincides with the cooler, nongrowing season when conditions are most favorable for recharge. Mundorff[7] estimates that recharge may amount to 600 mm in areas underlain by highly permeable young lavas that receive abundant precipitation. Considerable recharge also occurs by infiltration of water from streams that flow onto the plateau from the adjoining mountains. These sources of natural recharge are supplemented in agricultural areas by the infiltration of irrigation water.

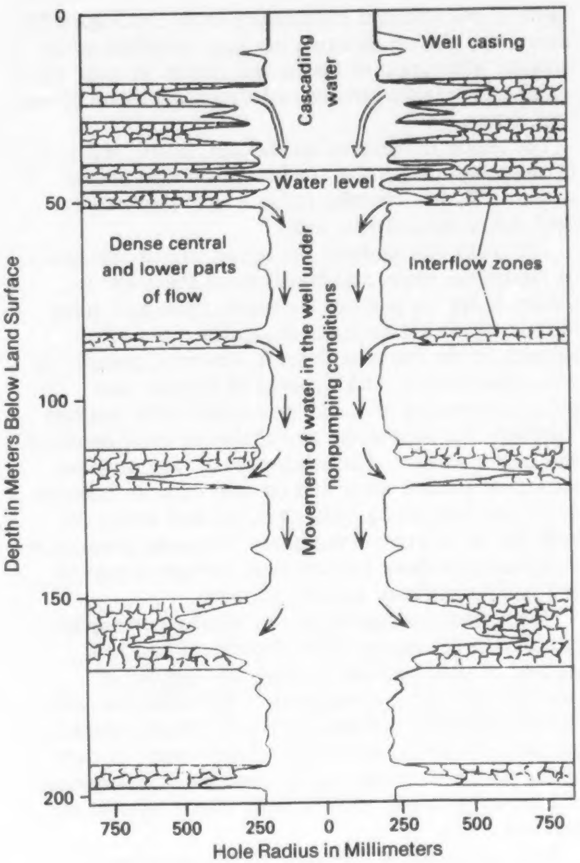
0
50
100
150
200
Depth in Meters Below Land Surface
Well casing
Cascading water
Water level
Dense central and lower parts of flow
Interflow zones
Movement of water in the well under nonpumping conditions
750
500
250
0
250
500
750
Hole Radius in Millimeters

Figure 78. Well in a Recharge Area in the Columbia River Group (Modified from Luzier and Burt, 1974.)[8]

Discharge from the ground-water system occurs as seepage to streams, as spring flow, and by evapotranspiration in areas where the water table is at or near the land surface. The famous Thousand Springs and other springs along the Snake River canyon in southern Idaho are, in fact, among the most spectacular displays of ground-water discharge in the world.

The large withdrawal of water in the Columbia Lava Plateau for irrigation, industrial, and other uses has resulted in declines in ground-water levels of as much as 30 to 60 m in several areas. In most of these areas, the declines have been slowed or stopped through regulatory restrictions or other changes that have reduced withdrawals. Declines are still occurring, at rates as much as a few meters per year, in a few areas.

4. Colorado Plateau and Wyoming Basin

(Thin soils over consolidated sedimentary rocks)

The Colorado Plateau and Wyoming Basin region occupies an area of 414,000 km^2 in Arizona, Colorado, New Mexico, Utah, and Wyoming. It is a region of canyons and cliffs of thin, patchy, rocky soils, and of sparse vegetation adapted to the arid and semiarid climate. The large-scale structure of the region is that of a broad plateau standing at an altitude of 2,500 to 3,500 m and underlain by horizontal to gently dipping layers of consolidated sedimentary rocks. As Figure 79 shows, the plateau structure has been modified by an irregular alternation of basins and domes, in some of which major faults have caused significant offset of the rock layers.

The region is bordered on the east, north, and west by mountain ranges that tend to obscure its plateau structure. It also contains rather widely scattered extinct volcanoes and lava fields.

The rocks that underlie the region consist principally of sandstone, shale, and limestone of Paleozoic to Cenozoic age. In parts of the region these rock units include significant amounts of gypsum (calcium sulfate). In the Paradox Basin in western Colorado the rock units include thick deposits of sodium- and potassium-bearing minerals, principally halite (sodium chloride). The sandstones and shales are most prevalent and most extensive. The sandstones are the principal sources of ground water and contain water in fractures developed both along bedding planes and across the beds and in interconnected pores. The most productive sandstones are those that are only partially cemented and retain significant primary porosity.

Unconsolidated deposits are of relatively minor importance in this region. Thin deposits of alluvium capable of yielding small to moderate supplies of ground water occur along parts of the valleys of major streams, especially adjacent to the mountain ranges in the northern and eastern parts of the region. In most of the remainder of the region there are large expanses of exposed bedrock, and the soils, where present, are thin and rocky.

Recharge of the sandstone aquifers occurs where they are exposed above the cliffs and in the ridges. Average precipitations ranges from about 150 mm in the lower areas to about 1,000 mm in the higher mountains. The heaviest rainfall occurs in the summer in isolated, intense thunderstorms during which some recharge occurs where intermittent streams flow across sandstone outcrops. However, most recharge occurs in the winter during snowmelt periods. Water moves down the dip of the beds away from the recharge areas to discharge along the channels of major streams through seeps and springs and along the walls of canyons cut by the streams.

The quantity of water available for recharge is small, but so are the porosity and the transmissivity of most of the sandstone aquifers. The water in the sandstone aquifers is unconfined in the recharge areas and is confined downdip. Because most of the sandstones are consolidated, the storage coefficient in the confined parts of the aquifers is very small. Even small rates of withdrawal cause extensive cones of depression around pumping wells.

The Colorado Plateau and Wyoming Basin is a dry, sparsely populated region in which most water supplies are obtained from the perennial streams. Less than 5 percent of the water needs are supplied by ground water, and the development of even small ground-water supplies requires the application of considerable knowledge of the occurrence of both rock units and their structure, and of the chemical quality of the water. Also, because of the large surface relief and the dip of the aquifers, wells even for domestic or small livestock supplies must penetrate to depths of a few hundred meters in much of the area. Thus, the development of ground-water supplies is far more expensive than in most other parts of the country. These

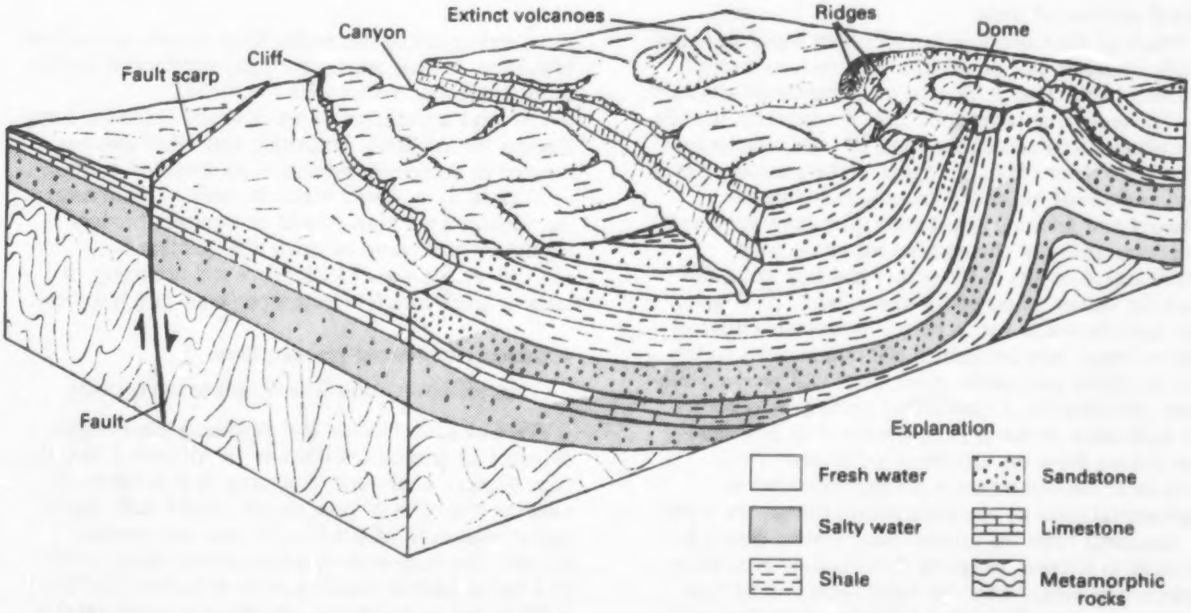

Figure 79. Topographic and Geologic Features of the Colorado Plateau and Wyoming Basin Region

negative aspects notwithstanding, ground water in the region can support a substantial increase over the present withdrawals.

As in most other areas of the country underlain by consolidated sedimentary rock, mineralized (saline) water—that is, water containing more than 1,000 mg/l of dissolved solids—is widespread. Most of the shales and siltstones contain mineralized water throughout the region and below altitudes of about 2,000 m. Freshwater—water containing less than 1,000 mg/l of dissolved solids—occurs only in the most permeable sandstones and limestones. Much of the mineralized water is due to the solution of gypsum and halite. Although the aquifers that contain mineralized water are commonly overlain by aquifers containing freshwater, this situation is reversed in a few places where aquifers containing mineralized water are underlain by more permeable aquifers containing freshwater.

5. High Plains

(Thick alluvial deposits over fractured sedimentary rock)

The High Plains region occupies an area of 450,000 km^2 extending from South Dakota to Texas. The plains are a remnant of a great alluvial plain built in Miocene time by streams that flowed east from the Rocky Mountains. Erosion has removed a large part of the once extensive plain, including all of the part adjacent to the mountains, except in a small area in southeastern Wyoming.

The original depositional surface of the alluvial plain is still almost unmodified in large areas, especially in Texas and New Mexico, and forms a flat, imperceptibly eastward-sloping tableland that ranges in altitude from about 2,000 m near the Rocky Mountains to about 500 m along its eastern edge. The surface of the southern High Plains contains numerous shallow circular depressions, called playas, that intermittently contain water following heavy rains. As Figure 80 shows, other significant topographic features include sand dunes, which are especially prevalent in central and northern Nebraska, and wide, downcut valleys of streams that flow eastward across the area from the Rocky Mountains.

The High Plains region is underlain by one of the most productive and most extensively developed aquifers in the United States. The alluvial materials derived from the Rocky Mountains, which are referred to as the Ogallala Formation, are the dominant geologic unit of the High Plains aquifer. The Ogallala ranges in thickness from a few meters to more than 200 m and consists of poorly sorted and generally unconsolidated clay, silt, sand and gravel.

Younger alluvial materials of Quarternary age overlie the Ogallala Formation of late Tertiary age in most parts of the High Plains. Where these deposits are saturated, they form a part of the High Plains aquifer; in parts of south-central Nebraska and central Kansas, where the Ogallala is absent, they comprise the entire aquifer. The Quarternary deposits are composed largely of material derived from the Ogallala and consist of

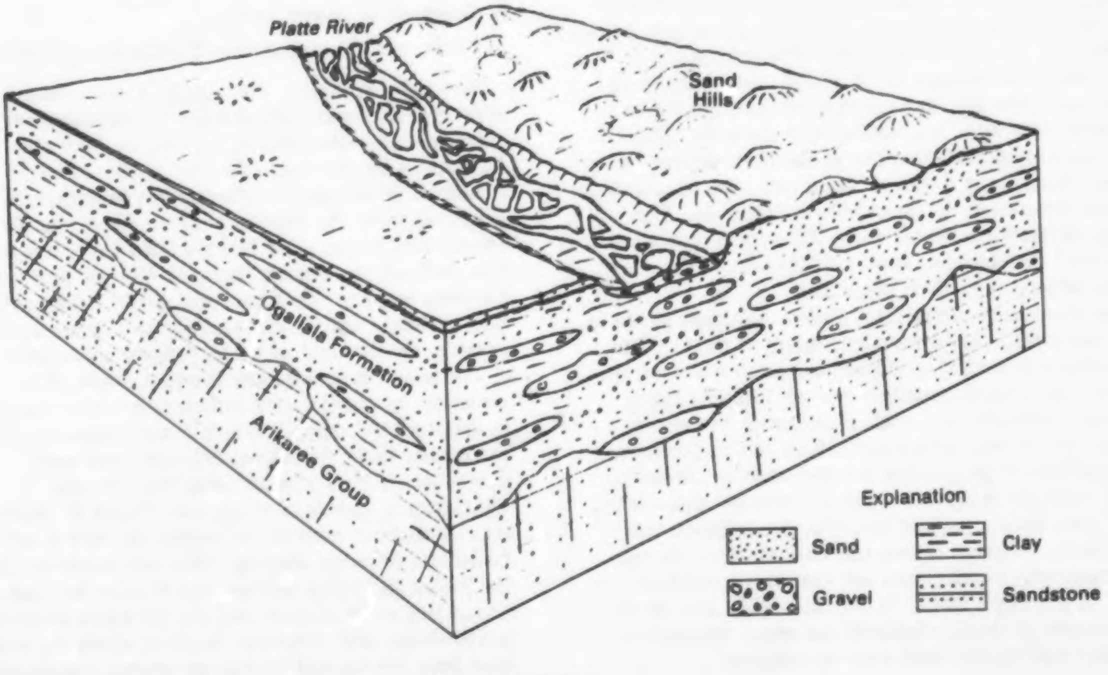

Figure 80. Topographic and Geologic Features of the High Plains Region.

gravel, sand, silt, and clay. An extensive area of dune sand occurs in the Sand Hills area north of the Platte River in Nebraska.

Other, older geologic units that are hydrologically connected to the Ogallala include the Arikaree Group of Miocene age and a small part of the underlying Brule Formation. The Arikaree Group is predominantly a massive, very fine to fine-grained sandstone that locally contains beds of volcanic ash, silty sand, and sandy clay. The maximum thickness of the Arikaree is about 300 m, in western Nebraska. The Brule Formation of Oligocene age underlies the Arikaree. In most of the area in which it occurs, the Brule forms the base of the High Plains aquifer. However, in the southeastern corner of Wyoming and the adjacent parts of Colorado and Nebraska, the Brule contains fractured sandstones hydraulically interconnected to the overlying Arikaree Group; in this area the Brule is considered to be a part of the High Plains aquifer.

In the remainder of the region, the High Plains aquifer is underlain by several formations, ranging in age from Cretaceous to Permian and composed principally of shale, limestone, and sandstone. The oldest of these underlies parts of northeastern Texas, western Oklahoma, and central Kansas and contains layers of relatively soluble minerals including gypsum, anhydrite, and halite (common salt), which are dissolved by circulating ground water.

Prior to the erosion that removed most of the western part of the Ogallala, the High Plains aquifer was recharged by the streams that flowed onto the plain from the mountains to the west as well as by local precipitation. The only source of recharge now is local precipitation, which ranges from about 400 mm along the western boundary of the region to about 600 mm along the eastern boundary. Precipitation and ground-water recharge on the High Plains vary in an east-west direction, but recharge to the High Plains aquifer also varies in a north-south direction. The average annual rate of recharge has been determined to range from about 5 mm in Texas and New Mexico to about 100 mm in the Sand Hills in Nebraska. This large difference is explained by differences in evaporation and transpiration and by differences in the permeability of surficial materials.

In some parts of the High Plains, especially in the southern part, the near-surface layers of the Ogallala have been cemented with lime (calcium carbonate) to form a material of relatively low permeability called caliche. Precipitation on areas underlain by caliche soaks slowly into the ground. Much of this precipitation collects in playas that are underlain by silt and clay, with the result that most of the water evaporates. It is only during years of excessive precipitation that significant recharge occurs and this, as noted above, averages only about 5 mm per year in the southern part of the High Plains. In the Sand Hills area about 20 percent of the precipitation (or about 100 mm annually) reaches the water table as recharge.

Figure 81 shows that the water table of the High Plains aquifer has a general slope toward the east. Gutentag and Weeks[9] estimate, on the basis of the average hydraulic gradient and aquifer characteristics, that water moves through the aquifer at a rate of about 0.3 m (1 ft) per day.

Natural discharge from the aquifer occurs to streams, to springs and seeps along the eastern boundary of the plains, and by evaporation and transpiration in areas where the water table is within a few meters of the land surface. However, at present the largest discharge is probably through wells. The widespread occurrence of permeable layers of sand and gravel, which permit the construction of large-yield wells almost any place in the region, has led to the development of an extensive agricultural economy largely dependent on irrigation. Most of this water is derived from ground-water storage, resulting in a long-term continuing decline in ground-water levels in parts of the region of as much as 1 m per year.

The depletion of ground-water storage in the High Plains is a matter of increasing concern in the region. However, from the standpoint of the region as a whole, the depletion does not yet represent a large part of the storage that is available for use. Weeks and Gutentag[10] estimate, on the basis of a specific yield of 15 percent of the total volume of saturated material, that the available (usable) storage in 1980 was about 4×10^2 m^3 (3.3 billion acre-ft). Luckey, Gutentag, and Weeks[11] estimate that this is only about 5 percent less than the storage that was available at the start of withdrawals. However, in areas where intense irrigation has long been practiced, depletion of storage is severe.

6. Nonglaciated Central Region

(Thin regolith over fractured sedimentary rocks)

As Figure 82 shows, the Nonglaciated Central region is an area of about 1,737,000 km^2 extending from the Appalachian Mountains on the east to the Rocky Mountains on the west. The part of the region in eastern Colorado and northeastern New Mexico is separated from the remainder of the region by the High Plains region. The Nonglaciated Central region also includes the Triassic Basins in Virginia and North Carolina and the "driftless" area in Wisconsin, Minnesota, Iowa, and Illinois where glacial deposits, if present, are thin and of no hydrologic importance.

The region is geologically complex. Most of its underlain by consolidated sedimentary rocks that range in age from paleozoic to Tertiary and consist largely of sandstone, shale, limestone, dolomite, and conglomerate. A small area in Texas and western Oklahoma is underlain by gypsum. Figure 83 shows that throughout most of the region the rock layers are horizontal or gently dipping. Principal exceptions are the Valley and Ridge section, the Wichita and Arbuckle Mountains in Oklahoma, and the Ouachita Mountains in Oklahoma and Arkansas, in all of which the rocks have been folded and extensively faulted. As Figure 84

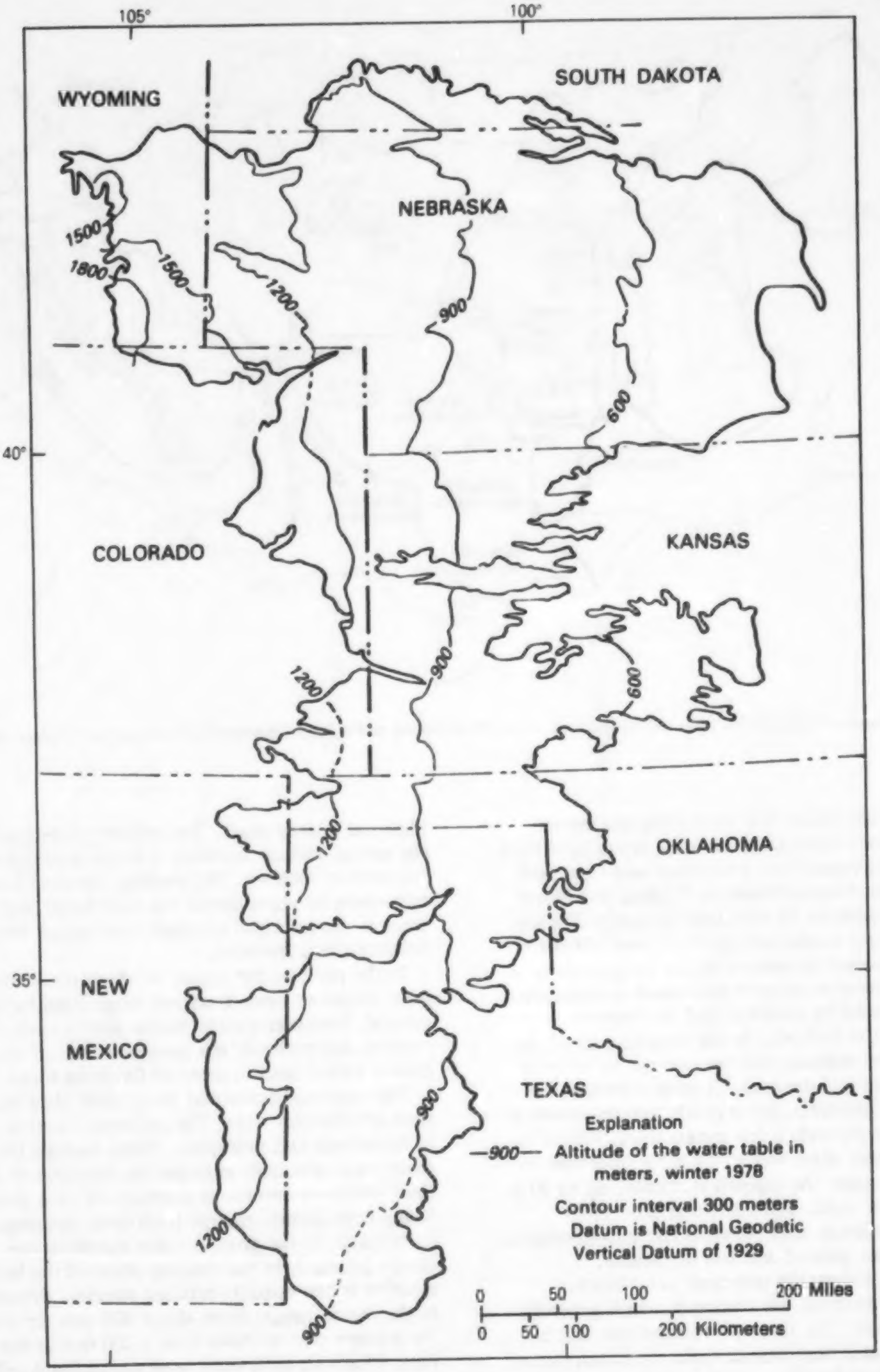

Figure 81. Altitude of the Water Table of the High Plains Aquifer.

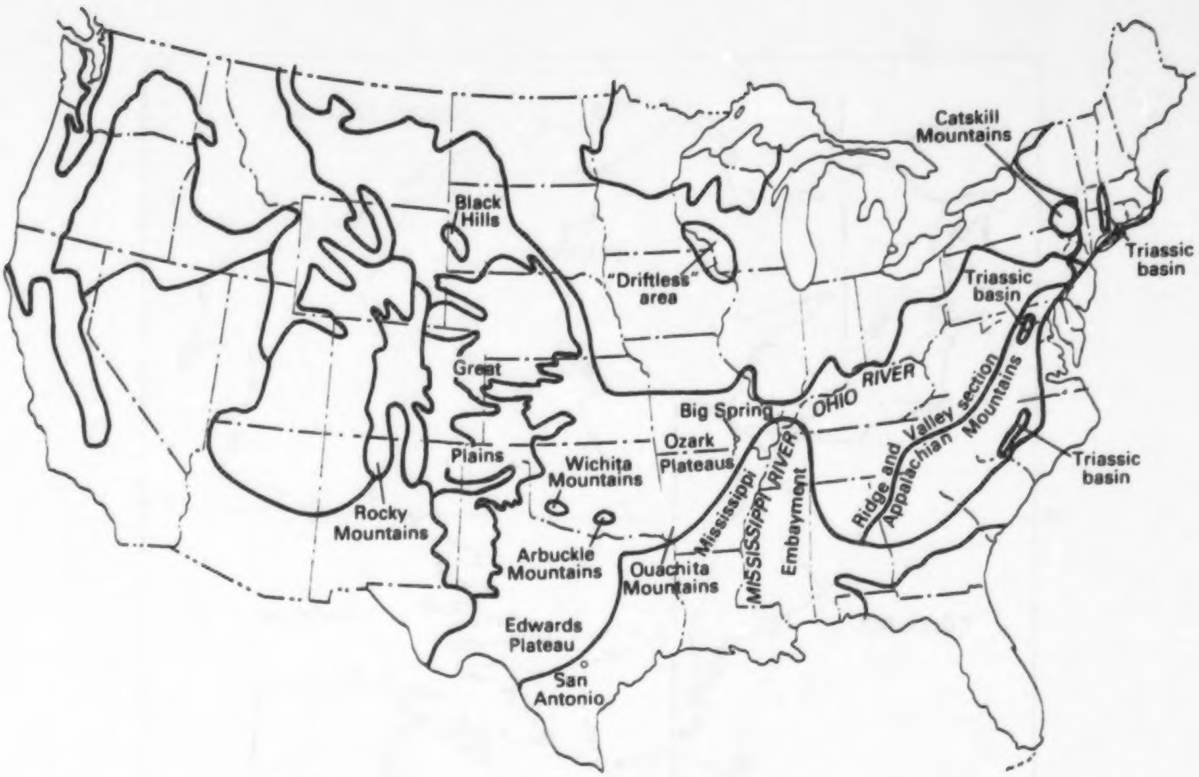

Figure 82. Location of Geographic Features Mentioned in the Discussions or Regions Covering the Central and Eastern Parts of the United States.

shows around the Black Hills and along the eastern side of the Rocky Mountains the rock layers have been bent up sharply toward the mountains and truncated by erosion. The Triassic Basins in Virginia and North Carolina are underlain by moderate to gently dipping beds of shale and sandstone that have been extensively faulted and invaded by narrow bodies of igneous rock.

The land surface in most of the region is underlain by regolith formed by chemical and mechanical breakdown of the bedrock. In the western part of the Great Plains the residual soils are overlain by or intermixed with wind-laid deposits. In areas underlain by relatively pure limestone, the regolith consists mostly of clay and is generally only a few meters thick. Where the limestones contain chert and in the areas underlain by shale and sandstone, the regolith is thicker, up to 30 m or more in some areas. The chert and sand form moderately permeable soils, whereas the soils developed on shale are finer grained and less permeable.

As Figure 83 shows, the principal water-bearing openings in the bedrock are fractures, which generally occur in three sets. The first set, and the one that is probably of greatest importance from the standpoint of ground-water as well yields, consists of fractures developed along bedding planes. The two remaining sets are essentially vertical and thus cross the bedding planes at a steep angle. The primary difference between the sets of vertical fractures is in the orientation of the fractures in each set. The vertical fractures facilitate movement of water across the rock layers and thus serve as the principal hydraulic connection between the bedding-plane fractures.

In the parts of the region in which the bedrock has been folded or bent, fractures range from horizontal to vertical. Fractures parallel to the land surface, where present, are probably less numerous and of more limited extent than in areas of flat-lying rocks.

The openings developed along most fractures are less than a millimeter wide. The principal exception occurs in limestones and dolomites. Water moving through these rocks gradually enlarges the fractures to form, in time, extensive cavernous openings or cave systems. Many large springs emerge from these openings.

Recharge of the ground-water system in this region occurs primarily in the outcrop areas of the bedrock aquifers in the uplands between streams. Precipitation in the region ranges from about 400 mm per year in the western part to more than 1,200 mm in the eastern part. This wide difference in precipitation is reflected in recharge rates, which range from about 5 mm per year in west Texas and New Mexico to as much as 500 mm per year in Pennsylvania and eastern Tennessee.

Discharge from the ground-water system is by springs and seepage into streams and by evaporation and transpiration.

The yield of wells depends on: (1) the number and size of fractures that are penetrated and the extent to which they have been enlarged by solution, (2) the rate of recharge, and (3) the storage capacity of the bedrock and regolith. Yields of wells in most of the region are small, in the range of 0.01 to 1 m^3 min^{-1} (about 2.5 to about 250 gallons per minute), making the Nonglaciated Central region one of the least favorable ground-water regions in the country. Even in parts of the areas underlain by cavernous limestone, yields are moderately low because of both the absence of a thick regolith and the large water-transmitting capacity of the cavernous openings which quickly discharge the water that reaches them during periods of recharge.

The exceptions to the small well yields are the cavernous limestones of the Edwards Plateau, the Ozark Plateaus, and the Ridge and Valley section. Figure 82

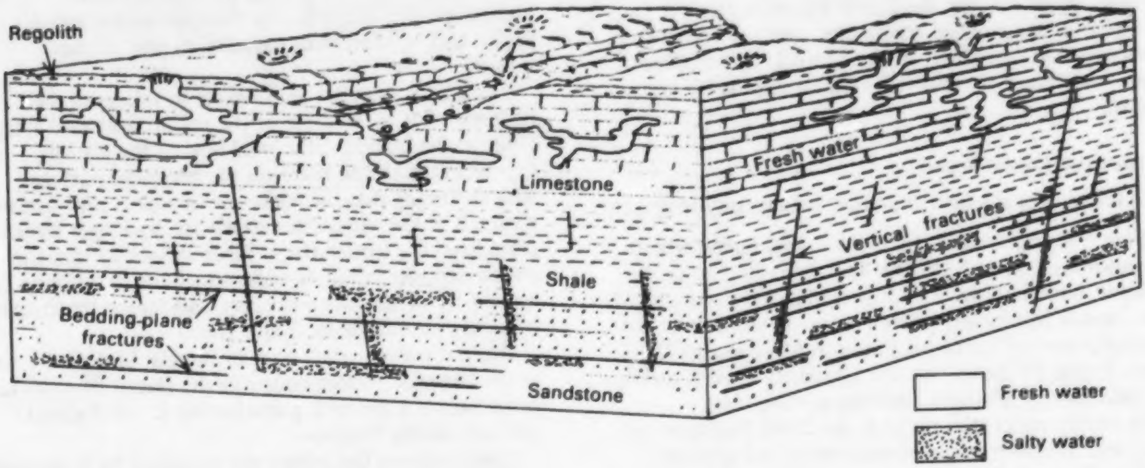

Figure 83. Topographic and Geologic Features of the Nonglaciated Central Region

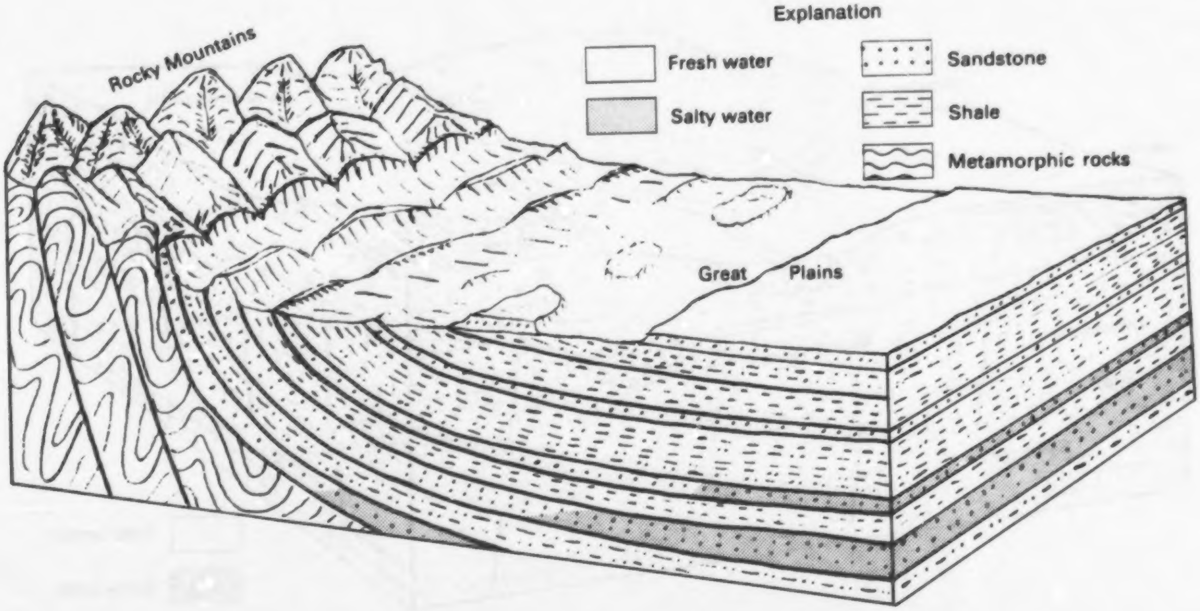

Figure 84. Topographic and Geologic Features Along the Western Boundary of the Nonglaciated Central Region.

shows the location of these areas. The Edwards Plateau in Texas is bounded on the south by the Balcones Fault Zone, in which limestone and dolomite up to 150 m in thickness has been extensively faulted, which facilitated the development of solution openings. This zone forms one of the most productive aquifers in the country. Wells of the City of San Antonio are located in this zone; individually, they have yields of more than 60 m^3min^{-1}.

As Figures 83 and 84 show, another feature that makes much of this region unfavorable for ground-water development is the occurrence of salty water at relatively shallow depths. In most of the Nonglaciated Central region, except the Ozark Plateaus, the Ouachita and Arbuckle Mountains, and the Ridge and Valley section, the water in the bedrock contains more than 1,000 mg/l of dissolved solids at depths less than 150 m.

7. Glaciated Central Region

(Glacial deposits over fractured sedimentary rocks)

Figure 82 shows the Glaciated Central region which occupies an area of 1,297,000 km^2 extending from the Triassic Basin in Connecticut and Massachusetts and the Catskill Mountains in New York on the east to the northern part of the Great Plains in Montana on the west. Figure 85 shows that the Glaciated Central region is underlain by relatively flat-lying consolidated sedimentary rocks that range in age from Paleozoic to Tertiary. The bedrock is overlain by glacial deposits which, in most of the area, consist chiefly of till, and unsorted mixture of rock particles deposited directly by the ice sheets. The till is interbedded with and overlain by sand and gravel deposited by meltwater streams, by silt and clay deposited in glacial lakes, and, in large parts of the North-Central States, by loess, a well-sorted silt believed to have been deposited primarily by the wind.

On the Catskill Mountains and other uplands in the eastern part of the region, the glacial deposits are typically only a few to several meters thick. In much of the central and western parts of the region, the glacial deposits exceed 100 m in thickness. The principal exception is the "driftless" area in Wisconsin, Minnesota, Iowa, and Illinois where the bedrock is overlain by thin soils. This area, both geologically and hydrologically, resembles the Nonglaciated Central region and is, therefore, included as part of that region.

The glacial deposits are thickest in valleys in the bedrock surface. In most of the region westward from Ohio to the Dakotas, the thickness of the glacial deposits exceeds the relief on the preglacial surface, with the result that the locations of valleys and stream channels in the preglacial surface are no longer discernible from the land surface. Figure 85 shows that the glacial deposits in buried valleys include, in addition to till and lacustrine silts and clays, substantial thicknesses of highly permeable sand and gravel.

Ground water occurs both in the glacial deposits and in the bedrock. Water occurs in the glacial deposits in pores between the rock particles and in the bedrock primarily along fractures.

Large parts of the region are underlain by limestones and dolomites in which fractures have been enlarged by solution. On the whole, caves and other large solution openings are much less numerous and hydrologically

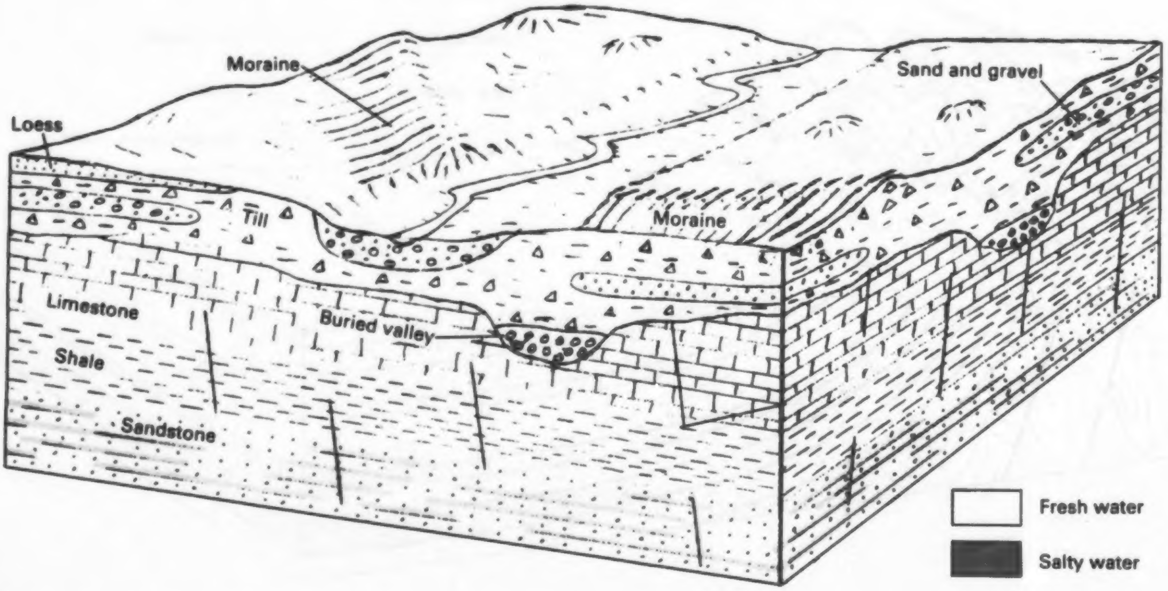

Figure 85. Topographic and Geologic Features of the Glaciated Central Region

much less important in the Glaciated Central region.

The glacial deposits are recharged by precipitation on the interstream areas and serve both as a source of water to shallow wells and as a reservoir for recharge to the underlying bedrock. Precipitation ranges from about 400 mm per year in the western part of the region to about 1,000 mm in the east. On sloping hillsides underlain by clay-rich till, the annual rate of recharge, even in the humid eastern part of the region, probably does not exceed 50 mm. In contrast, relatively flat areas underlain by sand and gravel may receive as much as 300 mm of recharge annually in the eastern part of the region.

Ground water in small to moderate amounts can be obtained any place in the region, both from the glacial deposits and from the bedrock. Large to very large amounts of ground water are obtained from the sand and gravel deposits and from some of the limestones, dolomites, and sandstones. The shales are the least productive bedrock formations in the region.

Because of the widespread occurrence of limestone and dolomite, water from both the glacial deposits and the bedrock contains as much as several hundred milligrams per liter of dissolved minerals and is moderately hard. Concentrations of iron in excess of 0.3 mg/l is a problem in water from some of the sandstone aquifers in Wisconsin and Illinois and locally in glacial deposits throughout the region. Sulfate in excess of 250 mg/l is a problem in water both from the glacial deposits and from the bedrock in parts of New York, Ohio, Indiana, and Michigan.

As is the case in the Nonglociated Central region mineralized water occurs at relatively shallow depth in bedrock in large parts of this region. The thickness of the freshwater zone in the bedrock depends on the vertical hydraulic conductivity of both the bedrock and the glacial deposits and on the effectiveness of the hydraulic connection between them. Both the freshwater and the underlying saline water move toward the valleys of perennial streams to discharge. As a result, the depth to saline water is less under valleys than under uplands. At depths of 500 to 1,000 m in much of the region, the mineral content of the water approaches that of seawater (about 35,000 mg/l). At greater depths, the mineral content may reach concentrations several times that of seawater.

8. Piedmont Blue Ridge Region

(Thick regolith over fractured crystalline and metamorphosed sedimentary rocks)

The Piedmont and Blue Ridge region is an area of about 247,000 km^2 extending from Alabama on the south to Pennsylvania on the north. The Piedmont part of the region consists of low, rounded hills and long, rolling, northeast-southwest trending ridges. The Blue Ridge is mountainous and includes the highest peaks east of the Mississippi.

The Piedmont and Blue Ridge region is underlain by bedrock of Precambrian and Paleozoic age consisting of igneous, and metamorphosed igneous, and sedimentary rocks. The land surface in the Piedmont and Blue

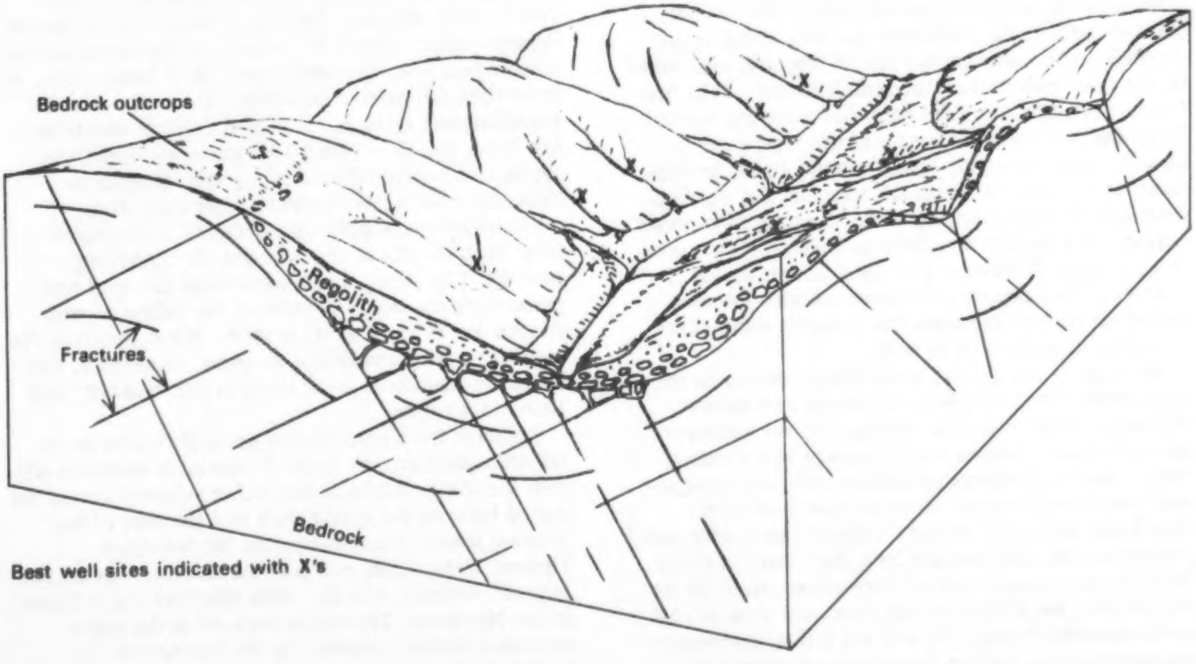

Figure 86. Topographic and Geologic Features of the Piedmont and Blue Ridge Region.

Ridge is underlain by clay-rich, unconsolidated material derived from in situ weathering of the underlying bedrock. This material, which averages about 10 to 20 m in thickness and may be as much as 100 m thick on some ridges, is referred to as saprolite. In many valleys, especially those of larger streams, flood plains are underlain by thin, moderately well-sorted alluvium deposited by the streams. While the distinction between saprolite and alluvium is not important, the term regolith is used to refer to the layer of unconsolidated deposits.

As Figure 86 shows the regolith contains water in pore spaces between rock particles. The bedrock, on the other hand, does not have any significant intergranular porosity. It contains water, instead, in sheetlike openings formed along fractures. The hydraulic conductivities of the regolith and the bedrock are similar and range from about 0.001 to 1 m day^{-1}. The major difference in their water-bearing characteristics is their porosities, the porosity of regolith being about 20 to 30 percent and the porosity of the bedrock about 0.01 to 2 percent. Small supplies of water adequate for domestic needs can be obtained from the regolith through large-diameter bored or dug wells. However, most wells, especially those where moderate supplies of water are needed, are relatively small in diameter and are cased through the regolith and finished with open holes in the bedrock. Although, the hydraulic conductivity of the bedrock is similar to that of the regolith, bedrock wells generally have much larger yields than regolith wells because, being deeper, they have a much larger available drawdown.

All ground-water systems function both as reservoirs that store water and as pipelines that transmit water from recharge areas to discharge areas. The yield of bedrock wells in the Piedmont and Blue Ridge region depends on the number and size of fractures penetrated by the open hole and on the replenishment of the fractures by seepage into them from the overlying regolith. Thus, the ground-water system in this region can be viewed, from the standpoint of ground-water development, as a terrane in which the reservoir and pipeline functions are effectively separated. Because of its larger porosity, the regolith functions as a reservoir that slowly feeds water downward into the fractures in the bedrock. The fractures serve as an intricate interconnected network of pipelines that transmit water either to springs or streams or to wells.

Recharge of the ground-water system occurs on the areas above the flood plains of stream, and natural discharge occurs as seepage springs that are common near the bases of slopes and as seepage into streams. With respect to recharge conditions, it is important to note that forested areas, which include most of the Blue Ridge and much of the Piedmont, have thick and very permeable soils overlain by a thick layer of forest litter. In these areas, even on steep slopes, most of the precipitation seeps into the soil zone, and most of this moves laterally through the soil and a thin, temporary, saturated zone to surface depressions or streams to discharge. The remainder seeps into the regolith below the soil zone, and much of this ultimately seeps into the underlying bedrock.

The Piedmont and Blue Ridge region has long been known as an area generally unfavorable for ground-water development. This reputation seems to have resulted both from the small reported yields of the numerous domestic wells in use in the region that were, generally, sited as a matter of convenience and from a failure to apply existing technology to the careful selection of well sites where moderate yields are needed. As water needs in the region increase and as reservoir sites on streams become incresingly more difficult to obtain, it will be necessary to make intensive use of ground water.

9. Northeast and Superior Uplands

(Glacial deposits over fractured crystalline rocks)

The Northeast and Superior Uplands region is made up of two separate areas totaling about 415,000 km^2. The Northeast Upland encompasses the Adirondack Mountains, the Lake Champlain valley, and nearly all of New England. The Superior Upland encompasses most of the northern parts of Minnesota and Wisconsin adjacent to the western end of Lake Superior.

Bedrock in the region ranges in age from Precambrian to Paleozoic, and as Figure 87 shows, consists mostly of intrusive igneous rocks and metamorphosed sedimentary rocks. Most have been intensively folded and cut by numerous faults.

As Figures 87 and 88 show, the bedrock is overlain by unconsolidated glacial deposits including till and gravel, sand, silt, and clay. The thickness of the glacial deposits ranges from a few meters on the higher mountains, which also have large expanses of barren rock, to more than 100 m in some valleys. The most extensive glacial deposit is till. In most of the valleys and other low areas, the till is covered by glacial outwash consisting of interlayered sand and gravel, ranging in thickness from a few meters to more than 20 m.

Ground-water supplies are obtained in the region from both the glacial deposits and the underlying bedrock. The largest yields come from the sand and gravel deposits, which in parts of the valleys of large streams are as much as 60 m thick. Water occurs in the bedrock in fractures similar in origin, occurrence, and hydraulic characteristics to those in the Piedmont and Blue Ridge region.

Recharge from precipitation generally begins in the fall after plant growth stops. It continues intermittently over the winter during thaws and culminated during the period between the spring thaw and the start of the growing season. Precipitation on the Northeast Upland, about 1,200 mm per year, is twice that on the Superior Upland, with the result that recharge is largest in the Northeast. The glacial deposits in the region serve as a storage reservoir for the fractures in the underlying bedrock.

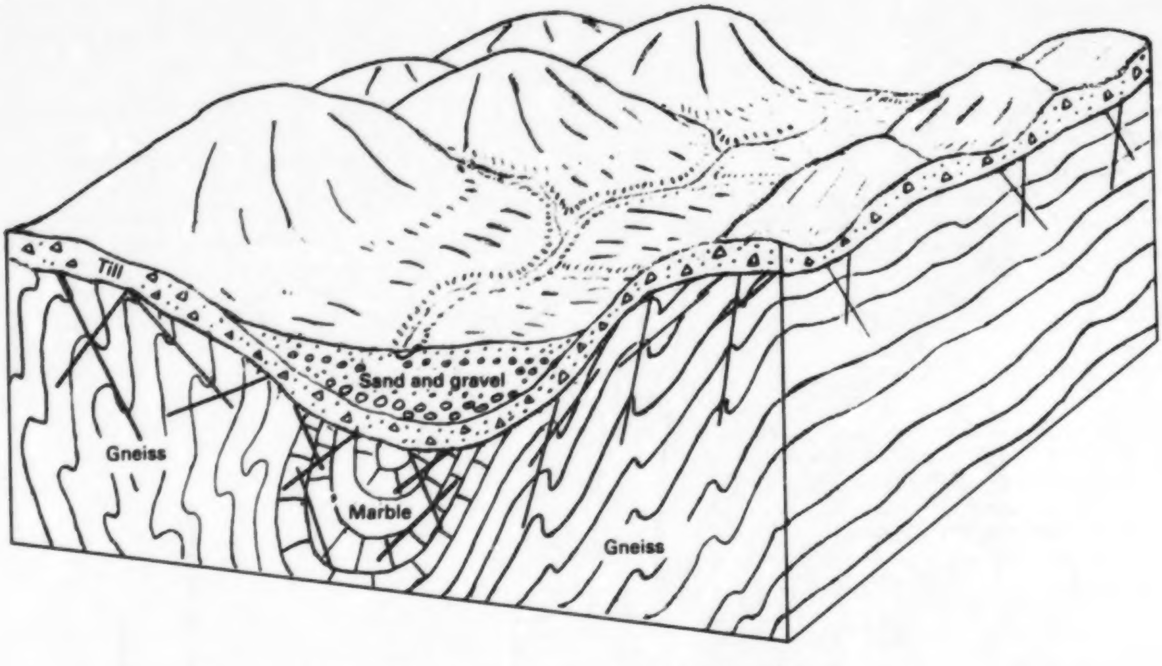

Figure 87. Topographic and Geologic Features on the Northeast and Superior Uplands Region.

Water supplies in the Northeast and Superior Uplands region are obtained from open-hole drilled wells in bedrock, from drilled and screened or open-end wells in sand and gravel, and from large-diameter bored or dug wells in till. The development of water supplies from bedrock, especially in the Superior Upland, is more uncertain than from the fractured rocks in the Piedmont and Blue Ridge region because the ice sheets that advanced across the region removed the upper, more fractured part of the rock and also tended to observe many of the fracture-caused depressions in the rock surface with the layer of glacial till.

Most of the rocks that underlie the Northeast and Superior Uplands are relatively insoluble, and consequently, the ground water in both the glacial deposits and the bedrock generally contains less than 500 mg/l of dissolved solids. Two of the most significant water-quality problems confronting the region, especially the Northeast Upland section, are acid precipitation and pollution caused by salts used to de-ice highways. Much of the precipitation falling on the Northeast in 1982 had a pH in the range of 4 to 6 units. Because of the low buffering capacity of the soils derived from rocks underlying the area, there is relatively little opportunity for the pH to be increased. One of the results of this is the gradual elimination of living organisms from many lakes and streams. The effect on ground-water quality, which will develop much more slowly, has not yet been determined. The second problem—that of de-icing salts—affects ground-water quality adjacent to streets and roads maintained for winter travel.

10. Atlantic and Gulf Coastal Plain

(Complexly interbedded sand, silt and clay)

The Atlantic and Gulf Coastal Plain region is an area of about 844,000 km^2 extending from Cape Cod, Massachusetts, to the Rio Grande in Texas. This region does not include Florida and parts of the adjacent States.

The topography of the region ranges from extensive, flat, coastal swamps and marshes, 1 to 2 m above sea level, to rolling uplands, 100 to 250 m above sea level, along the inner margin of the region.

The region is underlain by unconsolidated sediments that consist principally of sand, silt, and clay. These sediments, which range in age from Jurassic to the present, range in thickness from less than a meter near the inner edge of the region to more than 12,000 m in southern Louisiana. The sediments are complexly interbedded to the extent that most of the named geologic units into which they have been divided contain layers of the different types of sediment that underlie the region. These named geologic units dip toward the coast or toward the axis of the Mississippi embayment, with the result that those that crop out at the surface form a series of bands roughly parallel to the coast or to the axis of the embayment, as shown in Figure 89.

Although sand, silt, and clay are the principal types of material underlying the Atlantic and Gulf Coastal Plain, there are also small amounts of gravel interbedded with the sand, a few beds composed of mollusk shells, and small amounts of limestone present in the region. The most important limestone is the semi-

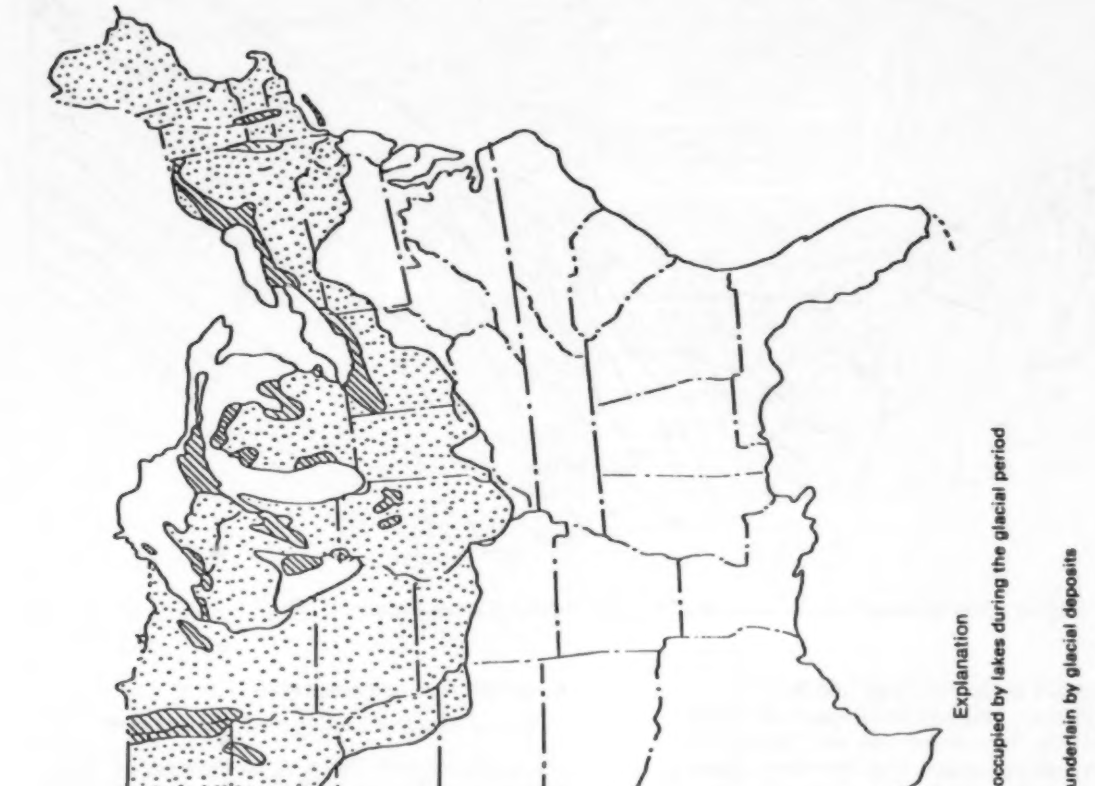
Explanation
Areas occupied by lakes during the glacial period
Areas underlain by glacial deposits
ALASKA

Figure 88. Glacial Features of the United States (Adapted from U.S. Geological Survey, 1970, p. 76.)[12]

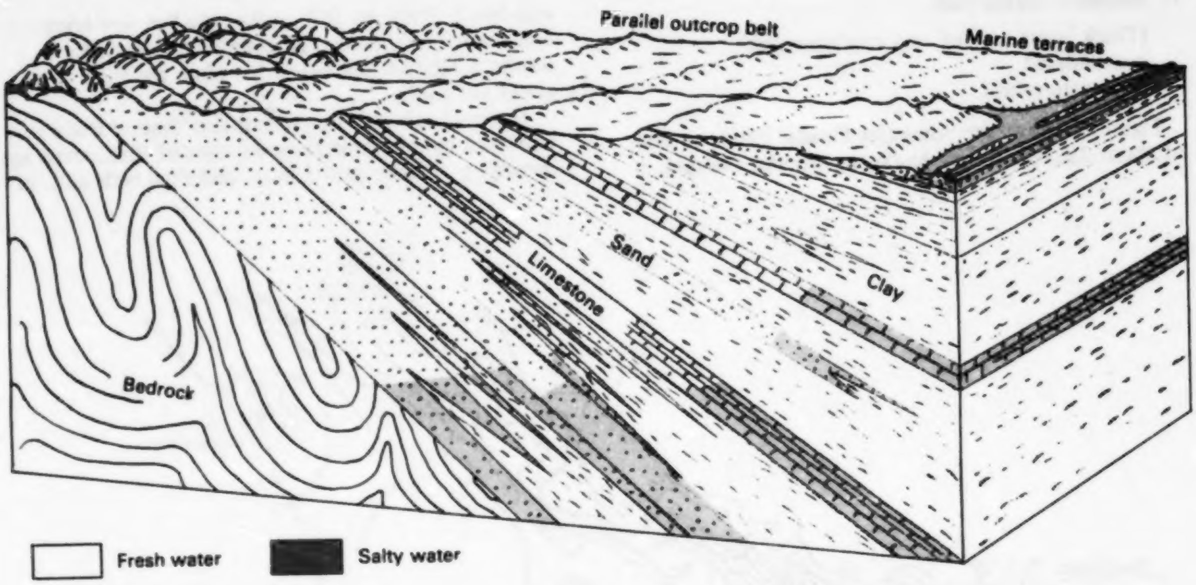

Figure 89. Topographic and Geologic Features of the Gulf Coastal Plain

consolidated Castle Hayne Limestone of Eocene age which underlies an area of about 26,000 km^2 in eastern North Carolina, is more than 200 m thick in much of the area, and is the most productive aquifer in North Carolina. A soft, clayey limestone (the chalk of the Selma Group) of Late Cretaceous age underlies parts of eastern Mississippi and western Alabama, but instead of being an aquifer, it is an important confining bed.

From the standpoint of well yields and ground-water use, the Atlantic and Gulf Coastal Plain is one of the most important regions in the country. Recharge to the ground-water system occurs in the interstream areas, both where sand layers crop out and by percolation downward across the interbedded clay and silt layers. Discharge from the system occurs by seepage to streams, estuaries, and the ocean.

Wells that yield moderate to large quantities of water can be constructed almost anywhere in the region. Because most of the aquifers consist of unconsolidated sand, wells require screens; where the sand is fine-grained and well sorted, the common practice is to surround the screens with a coarse sand or gravel envelope.

Withdrawals near the outcrop areas of aquifers are rather quickly balanced by increases in recharge and (or) reductions in natural discharge. Withdrawals at significant distances downdip do not appreciably affect conditions in the outcrop area and thus must be partly or largely supplied from water in storage in the aquifers and confining beds.

If withdrawals are continued for long periods in areas underlain by thick sequences of unconsolidated deposits, the lowered ground-water levels in the aquifer may result in drainage of water from layers of silt and clay. The depletion of storage in fine-grained beds results in subsidence of the land surface. Subsidence in parts of the Houston area totaled about 9 m as of 1978. Subsidence near pumping centers in the Atlantic Coastal Plain has not yet been confirmed but is believed to be occurring at a slower rate than along the Texas Gulf Coast.

Depletion of storage in the aquifers underlying large areas of the Atlantic and Gulf Coastal Plain is reflected in long-term declines in ground-water levels. These declines suggest that withdrawals in these areas are exceeding the long-term yield of the aquifers.

Another problem that affects ground-water development in the region concerns the presence of saline water in the deeper parts of most aquifers. In some of the deeper aquifers, the interface between freshwater and saltwater is inshore, but in parts of the region, including parts of Long Island, New Jersey, and Mississipi, the interface in the most intensively developed aquifers is a significant distance offshore. Pumping near the interfaces has resulted in local problems of saltwater encroachment.

Another significant feature of the ground-water system in this region is the presence of "geopressured" zones at depths of 1,800 to 6,100 m in Texas and Louisiana which contain water at a temperature of 80°C to more than 273°C. Water in these zones contains significant concentrations of natural gas, and the water in some zones is under pressures sufficient to support a column of water more than 4,000 m above land surface. Because the elevated temperature, natural gas, and high pressure are all potential energy sources, these zones are under intensive investigation.

11. Southeast Coastal Plain

(Thick layers of sand and clay over semi-consolidated
carbonate rocks)

Figure 90 shows the Southeast Coastal Plain, an area of about 212,000 km^2 in Alabama, Florida, Georgia, and South Carolina. It is a relatively flat, low-lying area. Much of the area, including the Everglades in southern Florida, is a nearly flat plain less than 10 m above sea level.

The land surface of the Southeast Coastal Plain is underlain by unconsolidated deposits of Pleistocene age consisting of sand, gravel, clay, and shell beds and, in

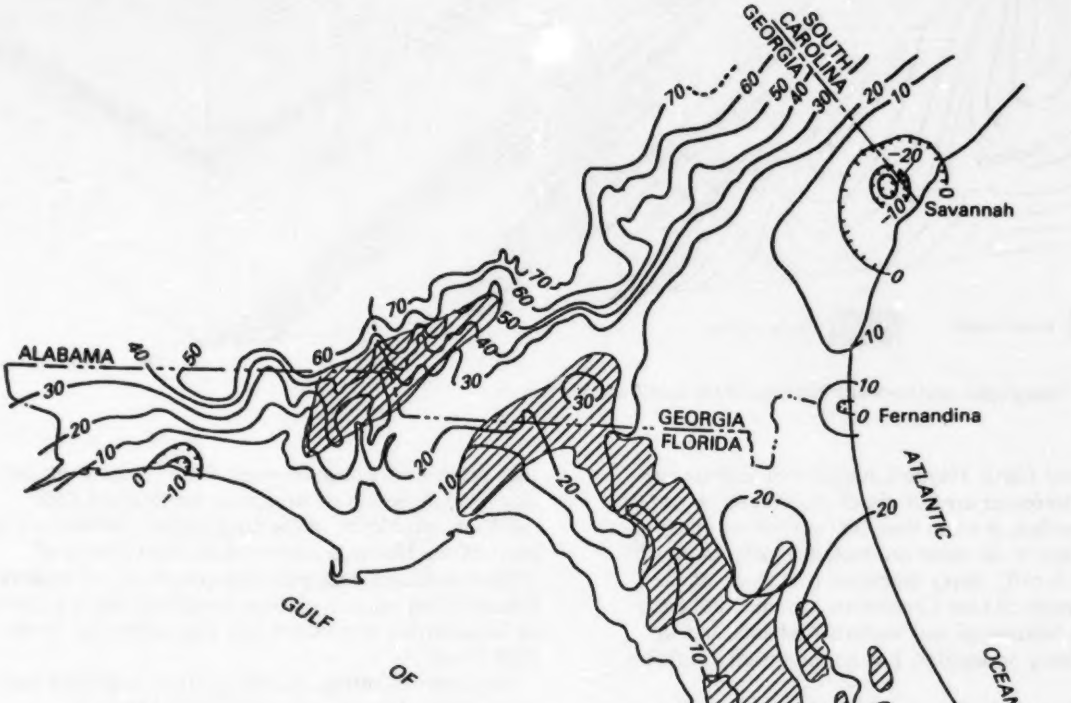

Figure 90. Potentiometric Surface for the Floridan Aquifer (Adapted from Johnston, Healy, and Hayes, 1981.)[13]

southeastern Florida, by semi-consolidated limestone. In most of the region, the surficial deposits rest on formations, primarily of middle to late Miocene age, composed of interbedded clay, sand, and limestone. The formations of middle to late Mioccene age or surficial deposits overlie semi-consolidated limestones and dolomites that are as much as 1,500 m thick.

The Tertiary limestone that underlies the Southeast Coastal Plain constitutes one of the most productive aquifers in the United States and is the feature that justifies treatment of the region separately from the remainder of the Atlantic and Gulf Coastal Plain. The aquifer, which is known as the Floridan aquifer, underlies all of Florida and southeast Georgia and small areas in Alabama and South Carolina. The Floridan aquifer consists of layers several meters thick composed largely of loose aggregations of shells and fragments of marine organisms interbedded with much thinner layers of cement and cherty limestone. The Floridan, one of the most productive acquifers in the world, is the principal source of ground-water supplies in the Southeast Coastal Plain region.

In southern Florida, south of Lake Okeechobee, and in a belt about 30 km wide northward along the east coast of Florida to the vicinity of St. Augustine, the water in the Floridan aquifer contains more than 100 mg/l of chloride. In this area, most water supplies are obtained from surfacial aquifers. The most notable of these aquifers underlies the southeastern part of Florida and, in the Miami area, consists of 30 to 100 m of cavernous limestone and sand and is referred to as the Biscayne aquifer. The Biscayne is an unconfined aquifer which is recharged by local precipitation and by infiltration of water from canals that drain water from impoundments developed in the Everglades. It is the principal source of water for municipal, industrial, and irrigation uses and can yield as much as 5 m^3 min^{-1} (1,300 gal min^{-1}) to small-diameter wells less than 25 m deep finished with open holes only 1 to 2 m long.

The surficial aquifers in the remainder of the region are composed primiarily of sand, except in the coastal zones of Florida where the sand is interbedded with shells and thin limestones. These surficial aquifers serve as sources of small ground-water supplies throughout the region and are the primary sources of ground water where the water in the Floridan aquifer contains more than about 250 mg/l of chloride.

The Floridan aquifer is the principal source of ground water in the region. Ground water in the upper part of the aquifer is unconfined in the principal recharge areas in Georgia and in west-central Florida, which are shown in Figure 90. In the remainder of the region, water in the aquifer is confined by clay in the Hawthorn Formation and in other beds that overlie the aquifer.

Recharge occurs where the potentiometric surface of

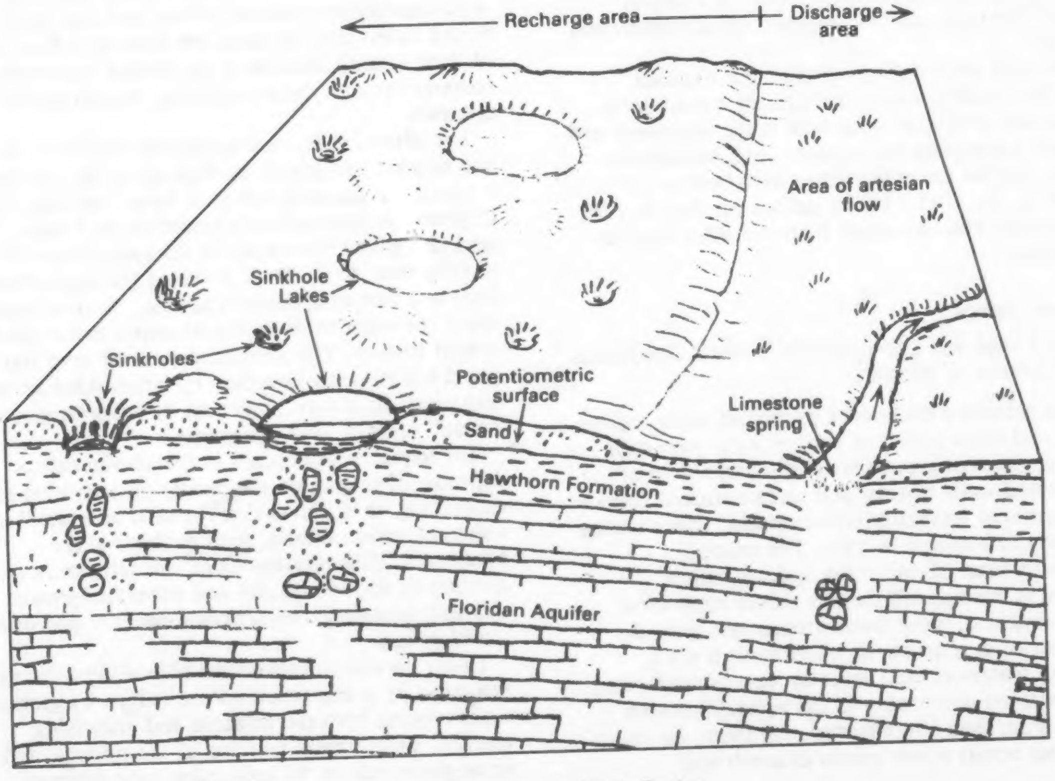

Figure 91. Topographic and Geologic Features of the Southeast Coastal Plain Region

the Floridan aquifer is lower than the water table in the overlying surficial aquifer. As Figure 90 shows, the principal recharge areas include a broad area along the west side of Florida extending from the central part of the peninsula to south-central Georgia and an area extending from west-central Florida through southeast Alabama into southwest Georgia. In these areas, recharge rates are estimated to exceed 120 mm yr^{-1} (5 in. yr^{-1}). Recharge occurs by infiltration of precipitation directly into the limestone, where it is exposed at the land surface, and by seepage through the permeable soils that partly mantle the limestone in the outcrop areas. Considerable recharge also occurs in the higher parts of the recharge areas through permeable openings in the confining beds, where these beds have been breached by the collapse of caverns in the limestone during the process of sinkhole formation. Figure 91 illustrates this sinkhole formation. Thus, the land surface in most of Florida north of Lake Okeechobee is marked by thousands of closed depressions ranging in diameter from a few meters to several kilometers. The larger depressions are occupied by lakes generally referred to as sinkhole lakes.

Discharge from the Floridan aquifer occurs through springs and by seepage to streams. Considerable discharge also occurs by diffuse seepage across the overlying confining beds in areas where the potentiometric surface of the aquifer stands at a higher altitude than the water table. In most of these areas wells open to the aquifer will flow at the land surface. The most spectacular discharge from the Floridan aquifer is through sinkholes exposed along streams and offshore.

Water supplies are obtained from the Floridan aquifer by installing casing through the overlying formations and drilling an open hole in the limestones and dolomites comprising the aquifer. Total withdrawals from the aquifer are estimated to have been about 13×10^6 m^3 day^{-1} (3.5 billion gallons per day) in 1978. Large withdrawals also occur from the other aquifers in the region.

12. Alluvial Valleys

(Thick sand and gravel deposits beneath floodplains and terraces of streams)

In the preceding discussions of ground-water regions, streams and other bodies of surface water were mentioned as places of ground-water discharge. In most areas ground-water systems and surface streams form a water system so intimately interconnected that a change in one causes a change in other. For example, withdrawals from ground-water systems reduce discharge to streams and thereby reduce streamflow. The movement of water from streams into ground-water systems in response to withdrawals is not a significant feature in most areas because ground-water withdrawals are dispersed over the uplands between streams rather than concentrated near them. An exception to this occurs where stream channels and floodplains are underlain by highly permeable deposits of sand and gravel. The large yields of these deposits, as well as the variability and availability of streamflow, encourage the development of these sand and gravel deposits as sources of ground water, and thus, encourage the concentration of withdrawals near streams.

From the standpoint of ground-water hydrology, three criteria are used to differentiate alluvial valleys from other valleys. These criteria are as follows:

1. The alluvial valleys contain sand and gravel deposits thick enough to supply water to wells at moderate to large rates. [Commonly, the water-transmitting capacity of the sand and gravel is at least 10 times larger than that of the adjacent (enclosing) rocks.]
2. The sand and gravel deposits are in hydraulic contact with a perennial stream that serves as a source of recharge and whose flow normally far exceeds that demand from any typical well field.
2. The sand and gravel deposit occurs in a clearly defined band ("channel") that normally does not extend beyond the floodplain and adjacent terraces. In other words, the width of the deposit is small or very small compared with its length.

According to these criteria, the valleys of streams that were not affected by glacial meltwater are not considered alluvial valleys. The floodplains in these valleys are commonly underlain only by thin deposits of fine-grained alluvium. These criteria also eliminate the "burried" valleys of the glaciated area. Although the water-transmitting capacity of the sand and gravel in buried valleys may be large, the yield to wells in most of them is small because of the limited opportunity for recharge through the surrounding, less-permeable materials.

The alluvial valleys are commonly underlain, in addition to sand and gravel, by deposits of silt and clay. In many of the glaciated valleys in New York and New England the land surface is underlain by a layer of sand and gravel that ranges in thickness from 1 to 2 m to more than 10 m. The bottom of this deposit ranges, from one part of a valley to another, from a position above the water table to several meters below the bottom of streams. This surficial deposit of sand and gravel is commonly underlain by interbedded silt and clay which is, in turn, underlain by a discontinuous "basal" layer of sand and gravel.

The sequence of deposits in the alluvial valleys depends, of course, on the history of deposition in the valleys. Figure 92 shows that the sand and gravel in the valleys of major streams, such as those of the Mississippi, Missouri, and Ohio, are commonly overlain by deposits of clay and other fine-grained alluvium deposited during floods since the end of the glacial period.

Under natural conditions the alluvial deposits are recharged by precipitation on the valleys, by ground water moving from the adjacent and underlying aquifers, by overbank flooding of the streams, and, in some glacial valleys, by infiltration from tributary

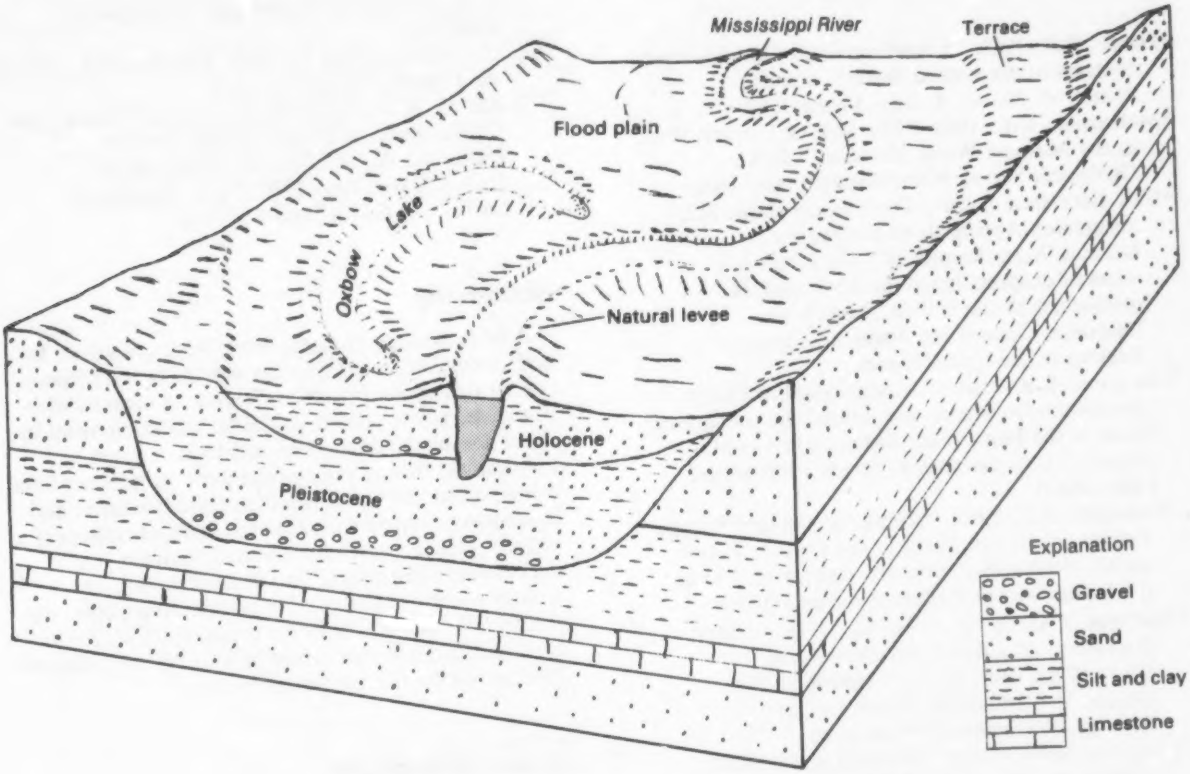

Figure 92. Topographic and Geologic Features of a Section of the Alluvial Valley of the Mississippi River

streams. Water in the alluvial deposits discharges to the streams in the valleys.

The layers of sand and gravel in the alluvial valleys are among the most productive aquifers in the country. They have been extensively developed as sources of water for municipalities, industries, and irrigation. Some of the gravel layers have hydraulic conductivities nearly as large as those of cavernous limestone. The large yields of the sand and gravel depend not only on their large water-transmitting capacity but also on their hydraulic connection to the streams flowing in the valleys. Large withdrawals from the deposits result in a reduction in ground-water discharge to the streams and, if large enough, cause infiltration of water from the streams into the deposits.

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A large number of publications were consulted, for both general and specific information, in the preparation of this paper. Specific reference to these publications generally is omitted in the text, both to avoid interruption of the discussions and to save space. Publications that served as primary references are listed below, under the categories of general references and references to regional discussions. General references include publications that were used both for background information on the classification of ground-water systems and for general information on the regions. References to the regional discussions include publications that served as a source of additional information on the individual regions.

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Ground Water-Surface Water Relationship

Introduction

The interrelations between ground water and surface water are of great importance in both regional and local hydrologic investigations and a wide variety of information can be obtained by analyzing streamflow data. Most commonly the surface water investigator deals with stream hydrographs, channel characteristics, geomorphology, or flood routing. Although the hydrogeologist may evaulate induced infiltration into a streamside aquifer, he is generally more interested in aquifer characteristics, such as hydraulic conductivity, thickness, boundaries, and well yields. Many hydrologists tend to ignore the fact that, at least in humid areas, ground-water runoff accounts for a significant part of a stream's total flow.

The evaluation of the ground-water component of runoff can provide important and useful information regarding regional recharge rates, aquifer characteristics, and ground-water quality, and can indicate areas of high potential yield to wells. The purpose of this chapter is to describe a number of techniques that can be used to evaluate runoff to obtain a better understanding and evaluation of ground-water resources. In particular the following will be examined:

1. Ground-water runoff
2. Surface runoff
3. Regional ground-water recharge rates
4. Determination of areas of relatively high permeability or water-yielding characteristics
5. Determination of the background concentration of ground-water quality
6. Estimation of evapotranspiration
7. Determination of the percentage of precipitation that is evaportranspired, becomes ground-water runoff, or becomes surface water runoff.

The approaches taken, admittedly some highly subjective, are based on: (1) short-term runoff events, (2) long-term hydrographs, and (3) dry-weather flow measurements. In the first approach a single event, such as a flood wave of a few hours or few days duration, can be analyzed, while the latter two approaches are based on annual stream hydrographs, flow-duration curves, or seepage runs. Short-term events may provide a considerable amount of information for a local area, while long-term events are most useful for regional studies.

Streamflow may consist of several components including ground-water runoff, surface runoff, effluent, and precipitation that falls directly into the channel.

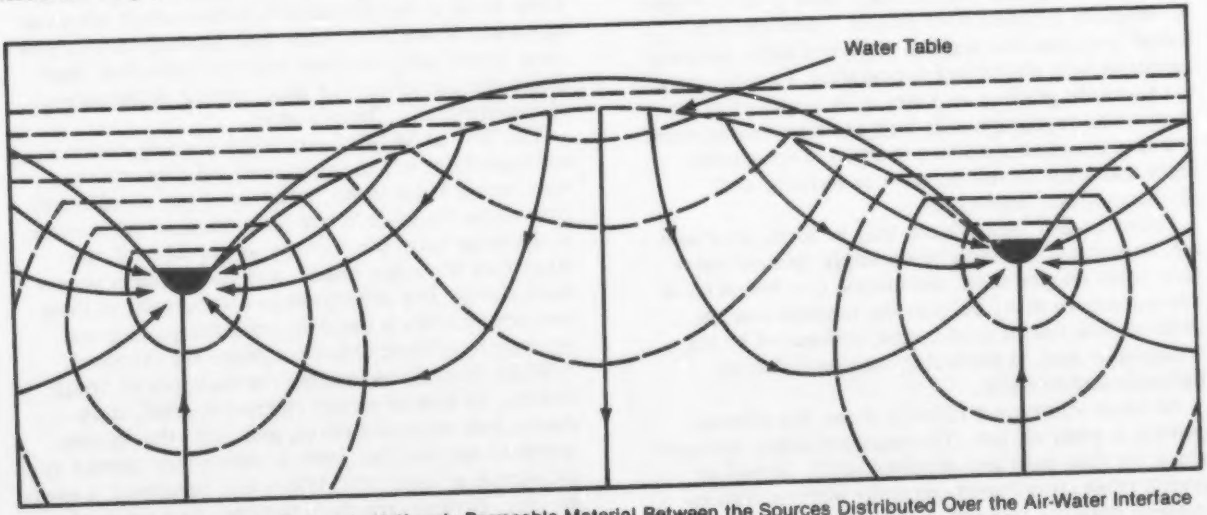


Figure 93. Approximate Flow Pattern in Uniformly Permeable Material Between the Sources Distributed Over the Air-Water Interface and The Valley Sinks (After Hubbert, 1940)

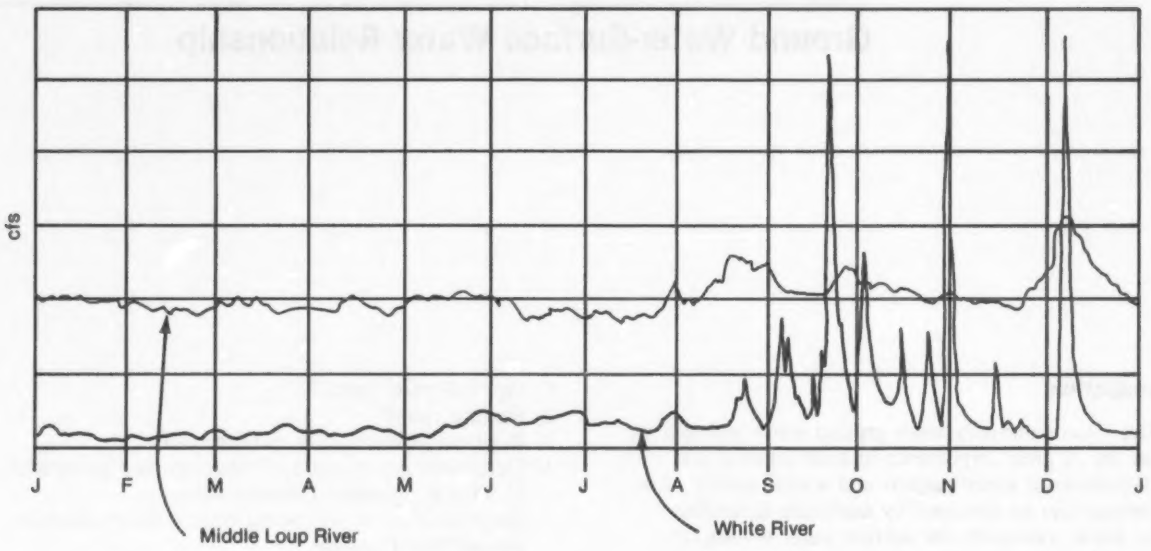


Figure 94. Hydrographs of Two Nearby Streams

The volume of water that is added by precipitation directly into the channnel is relatively small compared to the stream's total flow. The contribution by waste effluent may or may not be significant, since it depends on the activities that are occurring in the basin. In permeable basins in humid regions, ground-water runoff may account for 70 or 80 percent of the stream's annual discharge. The remainder is surface runoff, which originates as precipitation or snow melt that flows directly into the stream channel. This chapter is concerned largely with ground-water runoff and surface runoff and the separation of these two components.

In order to fully appreciate the origin and significance of ground-water runoff, it is first necessary to examine the regional ground-water flow system. Figure 93 illustrates a typical flow pattern. Particularly in humid and semi-arid regions, the water table generally conforms with the surface topography. Consequently, the hydraulic gradient or water table slopes away from divides and topographically high areas toward adjacent low areas, such as streams and rivers. Topographic highs and lows, therefore, serve as recharge and discharge areas, respectively.

Ground-water flow systems may be local, intermediate, or regional. As these terms imply, ground-water flow paths may be short, amounting to a few yards at one extreme to many miles in the regional case. Individual flow lines are, of course, influenced by the stratigraphy and, in particular, are controlled by hydraulic conductivity.

As water inflitrates a recharge areas, the mineral content is relatively low. The quality changes, however, along the flow path and dissolved solids, as well as several other constituents, generally increase with increasing distances traveled in the subsurface. It is for this reason that even nearby streams may be typified by different chemical quality. A stream, seep, or spring in a local discharge area may be less mineralized than that issuing from a regional discharge zone because of the increase in mineralization that takes place along longer flow paths. It must be remembered, however, that other conditions, such as soil type, solubility of the enclosing rocks, surface drainage characteristics, and waste disposal practices, may have a profound effect on water quality at any particular site.

Even streams in close proximity may differ considerably in discharge even though the size of the drainage area and climatic conditions are similar. Figure 94 gives the superimposed hydrographs of White River in southwestern South Dakota and the Middle Loup River in northwestern Nebraska which are good examples. White River has a low discharge throughout most of the year, but from May to September, flash floods are common. The wide extreme in discharge is characteristic of a flashy stream.

The flow of Middle Loup River is nearly constant, although from late spring to early fall higher flows may occur. These peaks, however, differ considerably from those found in White River because the increase in discharge takes place over a longer interval, the stage does not range widely, and the recession occurs more slowly. The differences in hydrographs of these two nearby rivers is puzzling, until the geology and topography of their respective basins are examined.

White River flows through the Badlands of South Dakota, an area of abrupt changes in relief, steep slopes, little vegetative cover, and rocks that consist largely of silt and clay, both of which may contain an abundance of bentonite. When wet, bentonite, a swelling clay, increases greatly in volume. As a result of these features, rainfall in the White River basin runs off quickly and there is little opportunity for infiltration and ground-water recharge to occur. Thus, intense rainstorms cause flash floods, such as those that occurred in June, August, and September.

The Middle Loup basin is carved into the undulating grassland topography of the sandhills of Nebraska where surficial materials consist of wind blown sand. Since the low relief, grass-covered surface promotes infiltration, precipitation is readily absorbed by the underlying sand. As a result, there is very little surface runoff and a great amount of infiltration and ground-water recharge. The ground water slowly migrates to the river channel, thus providing a high sustained flow. In a comparison of the hydrographs of these two rivers, it is evident that the geologic framework of the basin serves as a major control on runoff. This further implies that in any regional hydrologic study, the investigation should begin with an examination of geologic maps.

Gaining and Losing Streams

Although the discharge of most streams increases downstream, the flow of some streams diminishes. These streams are referred to as gaining or losing, respectively. The hydrologic system, however, is even more complex, because a stream that may be gaining in one season, may be losing during another. Furthermore, various activities of man may also affect a stream's discharge.

Under natural conditions a gaining stream is one where the water table is above the base of the stream channel. Of course the position of the water table fluctuates throughout the year in response to differences in ground-water recharge and discharge. Normally the water table is highest in the spring, which is the annual major period of ground-water recharge. From spring to fall, very little recharge occurs and the amount of ground-water in storage is slowly depleted as it seeps into the stream. Eventually, the water table may decline to the same elevation as a stream bottom, or even below it, at which time stream flow ceases except during periods of surface runoff. Following a period of recharge, caused either by infiltration of rainfall or seepage from a flood wave, the water table may again rise and temporarily contribute ground-water runoff.

Figure 95 shows a generalized diagram of the hydrology of a stream during two seasons of the year. During the spring, the water table is high and the gradient dips steeply towards the stream. If stream flow was measured at selected points, it would be found that the discharge increases downstream because of the addition of ground-water runoff. That is, it is a gaining stream. In the fall when the water table lies at or below the stream bottom, however, the same stream might become a losing stream. During a major runoff event the stage in the stream would be higher than the adjacent water table and water would migrate from the stream into the ground. The stream would continue to lose water until the water table and river stage were equal. When the stage declined, ground-water runoff would begin again.

In this case the stream changed from gaining to losing and back again to gaining. Similar situations may occur over longer intervals, such as during droughts. As a drought continues, the water table slowly declines as ground-water storage is depleted. A period of high flows, such as release from a dam, may cause tremen-

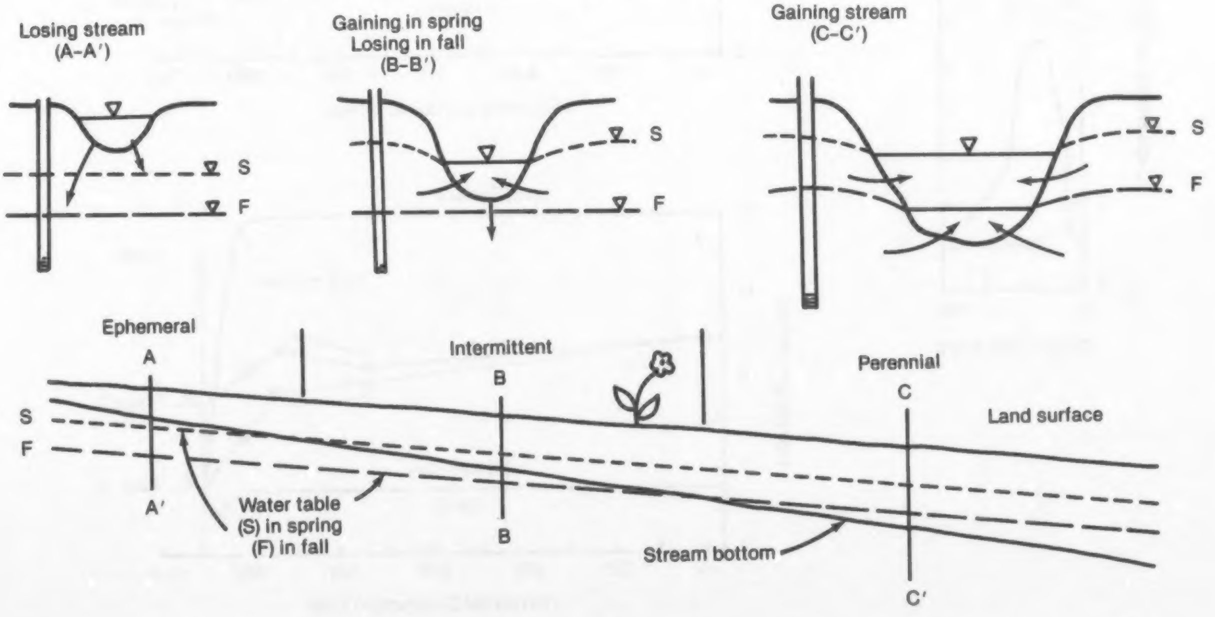


Figure 95. The Relation Between the Water Table and Stream Types

dous amounts of water to flow from the channel back into the ground, thus saturating the depleted streamside deposits. It may require weeks of high flow to replenish the ground-water reservoir and until this is accomplished, the stream will be losing.

Some streams, particularly in arid and karst regions, are nearly always losing. Examples include those channels that cross coarse-grained alluvial fans. Even during flash floods, the great mass of flood water soon spreads out over the fan or adjacent desert to inflitrate or evaporate.

Because of the extensive network of solution openings in karst terrain, the water table may consistently lie below the bottom of all the streams. During a period of runoff, the water may rapidly flow into sink holes and solution openings or simply disappear into a swallow hole in a stream channel, only to appear again perhaps several miles downstream.

Gaining and losing streams can also be created artifically. Where well fields lie along stream channels and induce water to flow from the stream to the well, stream flow is diminished. In some cases stream depletion by pumping wells has proceeded to such an extent that the stream channels are dry throughout the year. Conversely, in some irrigated regions, so much infiltration occurs that the water table rises to near land surface. The underlying soil and ground water may become highly mineralized by the leaching of soluble salts. These highly mineralized waters may discharge into a stream, increasing its flow but deteriorating the chemical quality. In other places, municipal or industrial wastes may add considerably to a stream's flow, also deteriorating its quality. In fact, at certain times of the year, the entire flow may consist of waste water.

Bank Storage

Figure 96 shows that, as a flood wave passes a particular stream cross-section, the water table may rise in the adjacent stream-side deposits. The rise is caused by two phenomena. First, the stream stage, which is higher than the water table, will temporarily block ground-water runoff, thus increasing the amount of

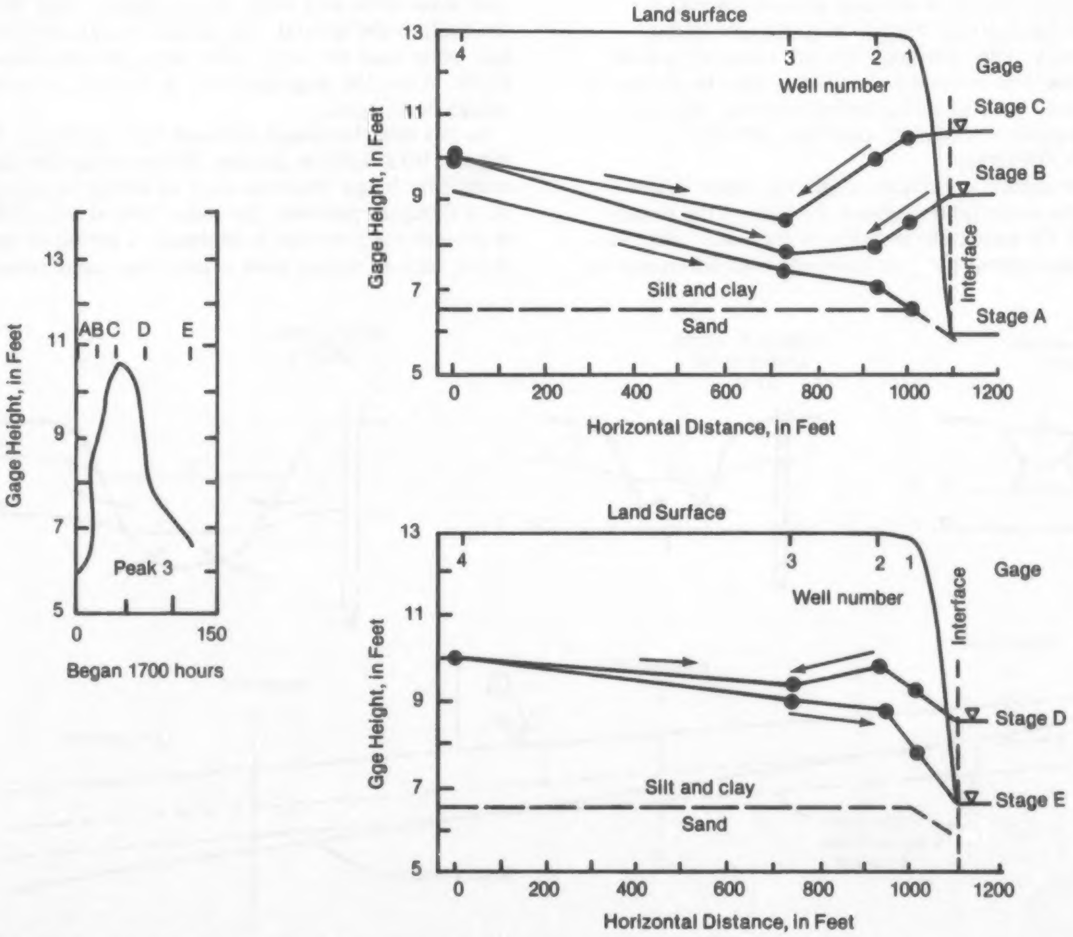


Figure 96. Movement of Water Into and Out of Bank Storage Along a Stream in Indiana

ground water in storage. Secondly, because of the increased head in the stream, water will flow from the channel into the ground, thus providing another component of water added to storage.

Once the flood waves begin to recede, which may occur quite rapidly, the newly added ground water begins to flow back into the channel, rapidly at first and then more slowly as the gradient decreases. This temporary storage of water in the near vicinity of the stream channel is called bank storage.

The rising and recession limbs of a hydrograph of a flood wave should provide clues concerning bank storage and stream-side permeability. For example, where stream-side deposits are of low permeability, such as clay or shale, the rising limb should be quite steep, but more gradual where the deposits are permeable. Since there would be little or no bank storage in the first case, recession curves also should be steep, but the release from bank storage in a permeable basin should reduce the slope of the recession curve.

Effect of the Geologic Framework on Stream Hydrographs

Unfortunately, the discharge of ground water into a stream is not always as simple as has been implied from the above examples. As Figure 97 shows, an examination of the aquifer framework and its effect on a stream hydrograph is enlightening. Notice in Figure 97a, that the stream channel is deeply cut into a shale that is overlain by sand. Ground water flows into the stream along a series of springs and seeps issuing at the sand-shale contact. During a runoff event the stream stage rises but, even at its peak, the stage remains below the top of the shale. In this case, the contribution of ground water remains constant despite the rise in stage. To separate the ground-water runoff component from the stream hydrograph, one merely needs to draw a straight line from the inflection points of the rising and falling limbs.

In Figure 97b the stream channel is cut into a deposit of sand that is underlain by shale. Ground water flows into the stream, but as the stage rises, ground-water runoff decreases and eventually stops. Surface water then begins to flow into the ground where it is retained as bank storage. As the stage declines, ground water again starts to discharge into the channel eventually providing the entire flow. This is the classic case of bank storage. Hydrograph separation is more difficult in this case.

Figure 97c is a combination of the previous two examples. Ground water from a perched aquifer contributes a steady flow, while bank storage is gained and then released from the stream-side aquifer. Hydrograph separation is even more difficult in this situation because of the contribution from both aquifers.

The final example, Figure 97d, consists of three aquifers—one perched, one in direct contact with the stream, and one deeper, artesian aquifer. As the stream rises, there is a decrease in the head difference between the stream and the artesian aquifer. The decrease in head difference will reduce upward leakage from the artesian aquifer, the amount depending on the thickness and vertical permeability of the confining bed and the head difference.

Single-Event Hydrograph Separation Techniques

Following a runoff event, the water held as bank storage begins to discharge into the channel. In the beginning the rate of bank storage discharge is high because of the steep water-level gradient, but as the gradient decreases so does ground-water runoff. The recession segment of the stream hydrograph gradually tapers off into what is called a depletion curve. To a large extent, the shape of the depletion curve is controlled by the permeability of the stream-side deposits, although soil moisture and evapotranspiration may play important roles.

Depletion Curves

Intervals between surface runoff events are generally short and for this reason, depletion curves are plotted as a combination of several arcs of the hydrograph with the arcs overlapping in their lower parts, as shown in Figure 98. To plot a depletion curve, tracing paper is placed over a hydrograph of daily flows and, using the same horizontal and vertical scales, the lowest arcs of the hydrographs are traced, working backward in time from the lowest discharge to a period of surface runoff. The tracing paper is moved horizontally until the arc of another hydrograph coincides in its lower part with the arc already traced; this arc is plotted on top of the first. The process is continued until all the available arcs are plotted on top of one another.

The upward curving parts of the individual arcs are disregarded because presumably they are affected by channel storage or surface runoff, or both. The resulting continuous arc is a mean or normal depletion curve that presumably represents the hydrograph that would result from ground-water runoff alone during a protracted dry period.

Even for the same stream, there may be appreciable differences in the shape of the depletion curve at different times of the year. This is largely due to evaporation, transpiration, and temperature effects. In cases such as these, a family of depletion curves may be constructed. One curve should represent winter when there is little or no evapotranspiration, another curve should represent the summer when evapotranspiration is at its maximum, and perhaps a third curve should be prepared to represent intermediate conditions.

Depletion curves are the basis for estimating ground-water runoff during periods of surface runoff. They also shed a great deal of light on the characteristics of a ground-water reservoir.

Hydrograph Separation

A flood hydrograph is a composite hydrograph consisting of surface runoff superimposed on ground-water

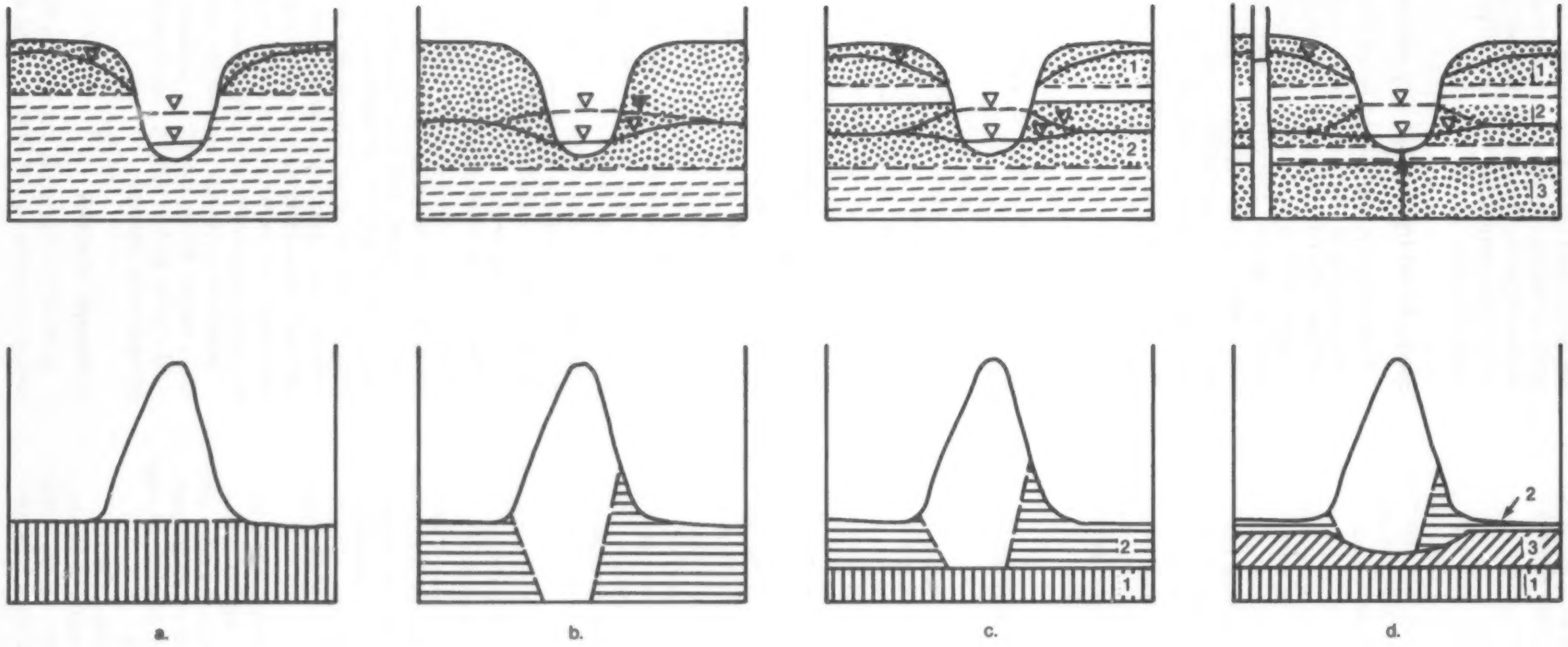


Figure 97. The Aquifer Framework in the Vicinity of a Stream Plays a Major Role in Ground-Water Runoff and Hydrograph Separation

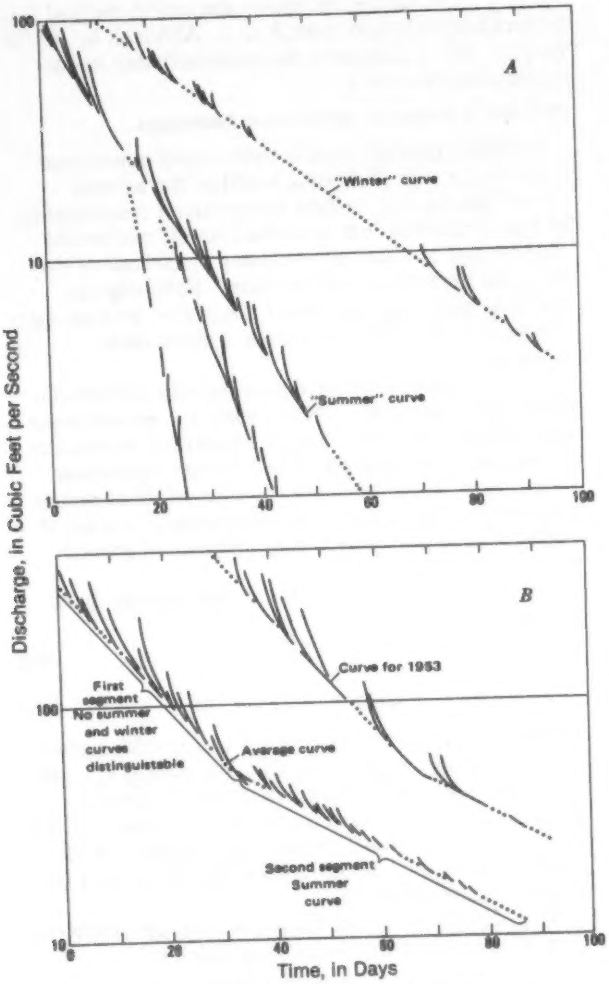


Figure 98. Ground-Water Depletion Curves have Different Shapes that Reflect the Seasons[14]

runoff. When attempting to separate these two components of flow, however, some problems generally occur. Whatever method is employed, there is always some question as to the accuracy of the division. One can only say that in any given case ground-water runoff is probably not less than about "x" or more than about "y". Keeping in mind the complexities of a runoff hydrograph brought about by variable parameters, and particularly the geology of the basin, an attempt will be made to develop some logical methods for hydrograph separation.

Using the flood hydrograph in Figure 99a, we can see that point A represents the start of surface runoff. Using a previously prepared depletion curve, the original recession can be extended to B. The area below AB represents the ground-water runoff that would have occurred had there been no surface runoff. Point D represents the end of surface runoff. A depletion curve can be matched with the recession limb, extending it from D to C. A partial envelope has now been formed that shows the upper and lower limits between which a line may reasonably be drawn to separate the two components of runoff. This assumption ignores possible effects brought about by difference in the geologic framework. This envelope forms a basis for the most commonly used separation methods which are described below.

Method 1. Using a depletion curve and starting at D in Figure 99b, extend the recession curve back to a line drawn vertically through the peak of the hydrograph (C). A second line is then extended from A, the start of surface runoff, to C. This method is more likely to be valid where ground-water runoff is relatively large and reaches the stream quickly.

Not uncommonly, the end of surface runoff is difficult to determine, but point D can be estimated by means of the equation

$$N = A^{0.2} \tag{48}$$

where N = the number of days after a peak when surface runoff ceases and A = the basin area in square miles. The distance N is measured directly on the hydrograph.

Method 2. In this example in Figure 99b, separation is accomplished merely by extending a straight line, originating at the start of surface runoff (A), to a point on the recession curve representing the end of surface runoff (D). This method of separation is certainly the simplist and is justifiable if little is known about the aquifer framework.

Method 3. In this example, also in Figure 99b, the pre-runoff recession line is extended from A to a point directly under the hydrograph peak (B). From this point a second line is projected to D, the end of surface runoff.

The separation technique to be employed should be based on knowledge of the hydrogeology of the basin, keeping in mind the affect of the geologic framework on the hydrograph.

Separation of Complex Hydrographs

Commonly runoff events occur at closely spaced intervals and there is insufficient time for the recession curve to develop before runoff again increases. This complicates hydrograph separation.

Figure 99c shows two methods that can be used to determine ground-water runoff under a complex hydrograph, which represents two storms.

Method 1. The recession curve preceding the first runoff event is continued to its intersection with a line drawn through the first peak (A-B). The distance N is calculated and measured. The recession limb of the first event is continued to its intersection with the N-days line (C-D). Line B-D is then constructed. The first recession trend is continued to its intersection with a line drawn through the peak of the second runoff event (C-D-E). From this point (E), the line is extended N days to F.

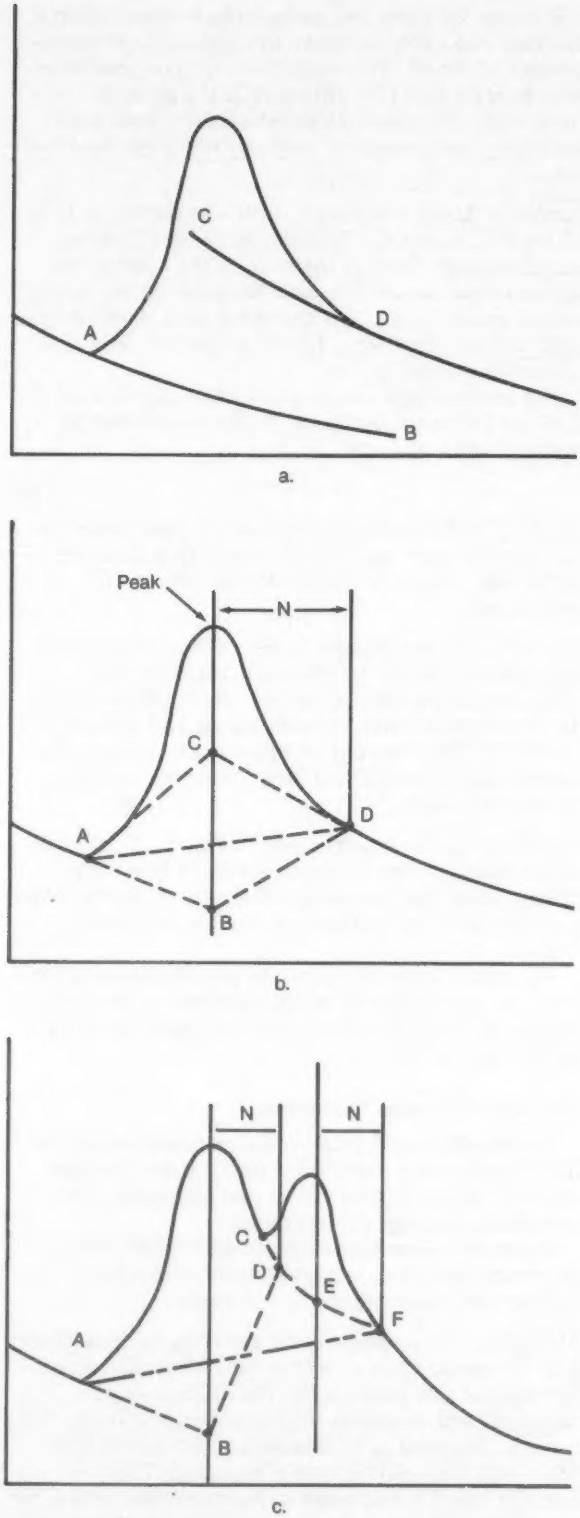


Figure 99. Separation of the Stream Hydrograph[15,16]

Method 2. As Figure 99c shows, the easiest method is to project a straight line from A to F. Although by far the simplest, this technique is not necessarily any less accurate than Method 1.

Hydrograph Separation by Chemical Techniques

Generally ground water is more highly mineralized than surface runoff. During baseflow the streams natural quality is at or near its maximum concentration of dissolved solids, but as surface runoff reaches the channel and provides an increasing percentage of the flow, the mineral content is diluted. Following the discharge peak, surface runoff diminishes, ground-water runoff increases, and the mineral content again increases.

Several investigators have used the relation between runoff and water quality to calculate the ground-water contribution from one or more aquifers or to measure streamflow. This method of hydrograph separation, which requires the solution of a series of simultaneous equations, is based on the concentration of a selected chemical parameter that is characteristics of ground water and surface runoff.

The basic equations, which may take several forms, are as follows:

$$Q_g + Q_s = Q \qquad (49)$$

$$C_gQ_g + C_sQ_s = CQ \qquad (50)$$

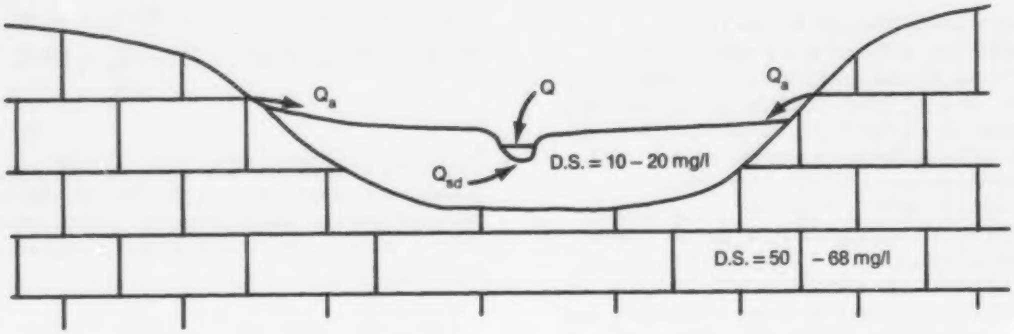
where Q_g, Q_s, and Q are ground-water runoff, surface runoff, and total runoff, respectively; and C_g, C_s, and C represent the concentration of dissolved mineral species or specific conductance of ground water, surface runoff, and total runoff, respectively. Usually specific conductance is used as the C parameter because of the relative ease of obtaining it.

If C_g, C_s, C, and Q are known we can determine the quantity of ground-water runoff as follows:

$$Q_g = Q\,(C - C_s)/(C_g - C_s) \qquad (51)$$

C_g is determined by measuring the specfic conductance in a well, in a series of wells, or during baseflow. The quality of surface runoff, C_s, is obtained from analysis of overland flow or, possibly in the case of small streams, at the period of peak discharge when the entire flow consists of surface runoff. It is assumed C_g and C_s are constant. C and Q are measured directly.

Visocky[17] used continuous recording equipment to measure specific conductance and stage (water level) in the Panther Creek Basin in north-central Illinois. By using the equations given above, he calculated the ground-water runoff component of the stream on the basis of the relationship between discharge and specific conductance. He also calculated and compared ground-water runoff as determined from a ground-water rating curve and found that the chemical method provided a lower estimate under normal conditions than did the rating curve technique. On the other hand, the chemical method indicated more ground-water runoff following storms that were preceeded by extended dry periods, which had caused considerable declines in water level in nearby observation wells.



$$Q_a + Q_{sd} = Q$$
$$CQ_a + CQ_{sd} = CQ$$

$$Q_a = ? \quad C_a = 50$$
$$Q_{sd} = ? \quad C_{sd} = 10$$
$$Q = 18 \quad C = 43$$

$$Q_a + Q_{sd} = 18$$
$$50\,Q_a + 10\,Q_{sd} = 43 \times 18$$

$$\begin{array}{rcl} -10Q_a - 10Q_{sd} & = & -180 \\ 50Q_a + 10Q_{sd} & = & 774 \\ \hline 40Q_a & = & 594 \\ & & Q_a = 14.85 \text{ fs} \end{array}$$

$$Q_a = \frac{Q\,(C - C_{sd})}{C_a - C_{sd}} \quad \text{OR} \quad Q_a = \frac{18(43 - 10)}{50 - 10} = \frac{594}{40} = 14.85 \text{ cfs}$$

Figure 100. Contribution to Econfina Creek During a Period of Dry Weather Flow When the Stream Discharge was 18 cfs and the Dissolved Solids Concentration was 43 mg/l:
From Sand Aquifer = 3.15 cfs
From Limestone Aquifer = 14.85 cfs

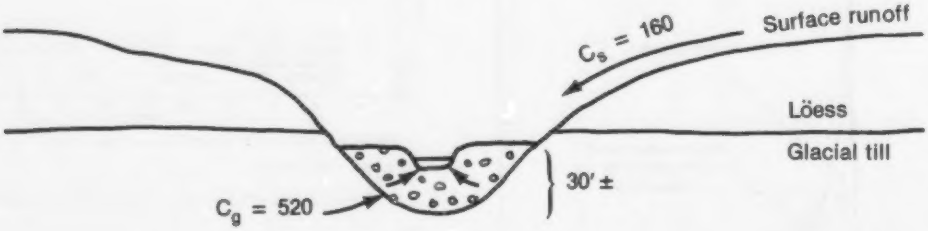


Figure 101. Four Mile Creek, Iowa

During baseflow, the quantity of ground-water discharge from surficial sand and from limestone in the Floridan artesian aquifer into Econfina Creek in northwest Florida was distinguished by Toler.[18] In this case, as Figure 100 shows, the artesian water had a dissolved solids content of 50-68 mg/1, while that from the surficial sand was only 10-20 mg/1. The artesian water discharged through a series of springs along the central part of the basin and amounted to 70 to 75 percent of the stream's baseflow. The equation used for this analysis is as follows:

$$Q_a = (C - C_{sd})/(C_a - C_{sd})\,Q \qquad (52)$$

where Q_a = artesian runoff, Q = runoff and C_{sd}, C_a, and C represent the dissolved solids in water from the sand, the artesian aquifer, and during any instant in the stream, respectively. Of course,

$$Q - Q_a = Q_{sd} \qquad (53)$$

Continuous streamflow and conductivity measurements were collected at a gaging station on Four Mile Creek in east-central Iowa by Kunkle.[19] The basin above the gage, which contains 19.5 square miles, consists largely of till that is capped on the uplands by loess. As Figure 101 shows, the stream lies in a valley that contains as much as 30 feet of permeable alluvium. Ground water from the alluvium and loess, as well as the stream during low flow, has an average specific conductance of 520 micromhos (C_g) while surface runoff is about 160 micromhos (C_s).

Figure 102 shows continuous record of discharge and conductivity representing a storm in September 1960. Instantaneous ground-water runoff during this event was calculated for several points under the hydrograph by using the following formulas:

$$Q_g + Q_s = Q \qquad (49)$$

$$C_gQ_g + C_sQ_s = CQ \qquad (50)$$

where Q_g = ground-water runoff, Q_s = surface runoff, Q = runoff and C_g, C_s, and C = specfic conductance of ground-water, surface runoff, and runoff, respectively. As determined from the graphs in Figure 102, where $Q = 2.3$ cfs, $C = 410$; $C_g = 520$ and $C_s = 160$, Q_g can be calculated as follows:

$$Q_g + Q_s = Q \qquad -160\,Q_g - 160\,Q_s = -368$$

$$520\,Q_g + 160\,Q_s = CQ \qquad \underline{520\,Q_g} + \underline{160\,Q_s} = \underline{943}$$

$$360_g \qquad 575$$

$$Q_g = 1.6\text{ cfs}$$

Therefore, when the stream's discharge (Q) was 2.3 cfs, ground-water runoff was 1.6 cfs. This calculation provides one point on the hydrograph. Several other points need to be determined so that a separation line can be drawn.

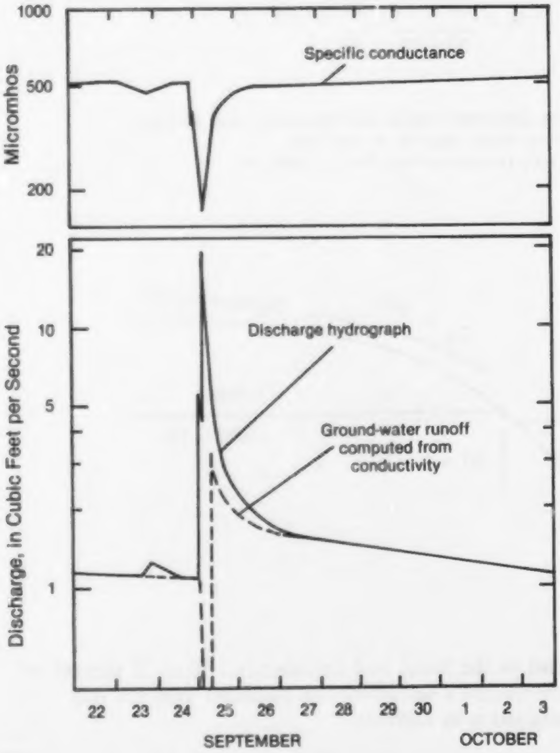


Figure 102. Hydrographs Showing Water Discharge, Specific Conductance, and Computed Ground-Water Runoff in Four Mile Creek near Traer, Iowa, September and October 1963.[19]

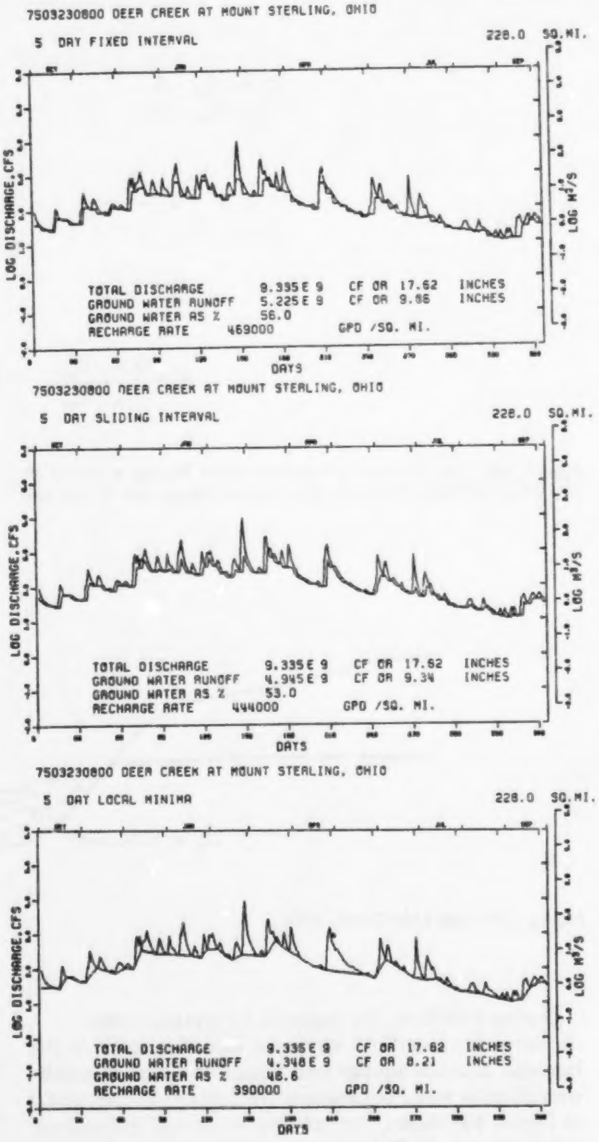


Figure 103. Deer Creek Hydrographs Separated by Three Methods and Statistical Data[20]

Computer Separation Programs

Various methods of hydrograph separation have been described, all of which are laborious, time consuming, and commonly, open to questions of accuracy and interpretation. In each case a mechanical technique is used to provide a number of points on a hydrograph through which a line can be drawn that separates ground-water runoff from surface runoff. Once this line is determined, one must measure, directly on the hydrograph, the daily components of streamflow and then sum the results.

Annual ground-water runoff divided by total discharge provides the percentage of streamflow that consists of ground water. Effective ground-water recharge is that quantity of precipitation that infiltrates, is not removed by evapotranspiration, and eventually discharges into a stream.

Effective ground-water recharge rates can be easily estimated with a computer program described by Pettyjohn and Henning.[20] This program separates the hydrograph by three different methods, provides monthly recharge rates and an annual rate, produces a flow-duration table, and gives the operator the option of generating with a line printer the separated hydrograph and a flow-duration curve; as illustrated in Figure 103. The data base is obtained from annual streamflow records, which are published by the U.S. Geological Survey. The computer program will operate on a mainframe or microcomputer.

Ground-Water Rating Curve

A widely used technique to measure streamflow is the surface water rating curve, which shows the relation between stage and discharge. Figure 104 shows a similar curve that shows the relation between the water table and streamflow called a ground-water rating curve. Prepared for those aquifer-stream systems that are hydrologically connected, the ground-water rating curve can be used to separate ground-water runoff from a stream hydrograph.

To prepare the curve, synchronous water table and stream discharge measurements are required. Ground-water levels are obtained either from: (1) a series of wells spread throughout the basin (2) a series of wells, each of which represents an area of similar geology, or (3) a single near-stream well. Wells influenced by pumping should not be used. If more than one well is used, water levels, referred to some datum such as sea level, must be averaged to form a composite hydrograph. Furthermore, measurements of both ground water and stream stage should be made only during rainless intervals when streamflow consists entirely of ground-water runoff. Selected water-level measurements are plotted on a graph with the mean daily streamflow and a smooth curve is drawn through the points.

The graph is used by determining, either from individual measurements or water-level recorder data, the ground-water stage, reading across the graph to the

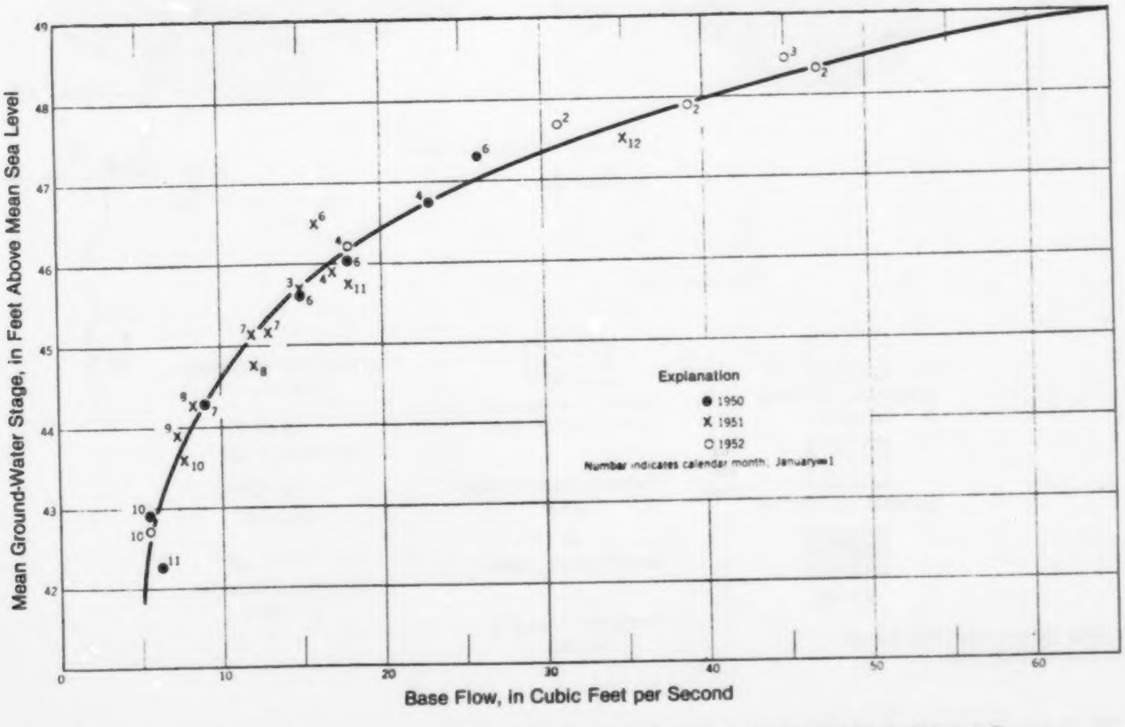
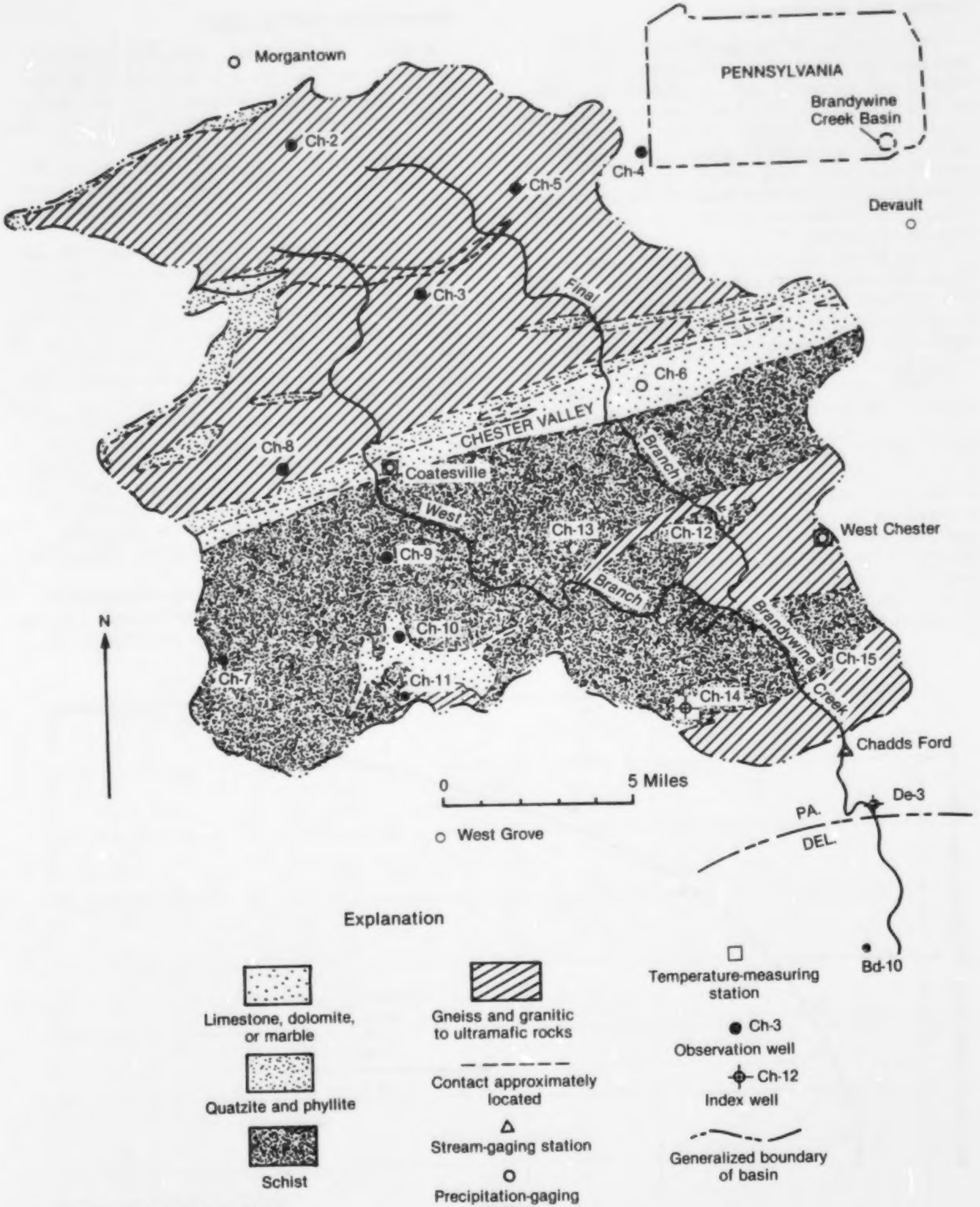


Figure 104. Rating Curve of Mean Ground-Water Stage Compared with Base Flow of Beaverdam Creek, Maryland[21]



Geology generalized from Bixom and Siese (1932, 1938)

Figure 105. Sketch Map of Brandywine Creek Basin, Showing Generalized Geology and Location of Hydrologic and Meteorologic Stations Used in Report[22]

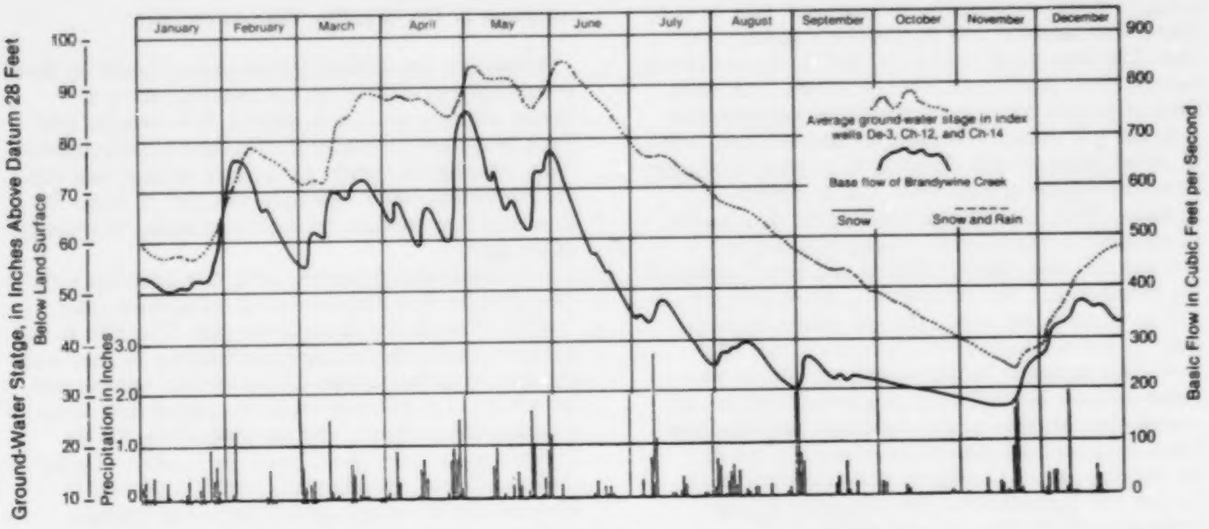


Figure 106. Composite Hydrograph of Three Index Wells and the Discharge of Brandywine Creek[22]

curve, and then reading down to the stream discharge. For example, in Figure 104 when the mean ground-water stage is 44.5 feet, ground-water runoff is 10 cfs. Any flow in excess of this amount is surface runoff. Daily values of ground-water runoff are plotted on the stream hydrograph, eventually forming a continuous line throughout the period of record.

Although wells produce only limited yields from crystalline rocks in the Piedmont Upland part of the Delaware River Basin, streams have unusually high base flows. Olmsted and Hely[22] used a ground-water rating curve to study this apparent inconsistency in a 287 square mile part of the Brandywine Creek basin in southeastern Pennsylvania, as illustrated in Figure 105.

Bedrock units in the dissected upland basin, which consists largely of folded Precambrian and Paleozoic igneous and metamorphic rocks, have similar hydrologic characteristics. Weathered material of variable thickness mantles the area and the water table lies largely within this zone. Precipitation averages about 44 inches.

The 16 observation wells used in this study ranged from 12 to 40 feet in depth; all tapped a weathered schist aquifer. Six or seven wells were measured weekly or immediately after storms and wells De-3, Ch-13, Ch-14 were selected as index wells. The average depth to water in all of the wells was 17.45 feet and the annual fluctuation was 5.75 feet.

Figure 106 shows a composite hydrograph of the three index wells and the discharge of Brandywine Creek. The curves have similar trends, differing only in amplitude following runoff events. This is to be expected because of the quick response time of a stream. Certainly the closer an observation well is to a stream, the more nearly the hydrographs will approach a similar shape.

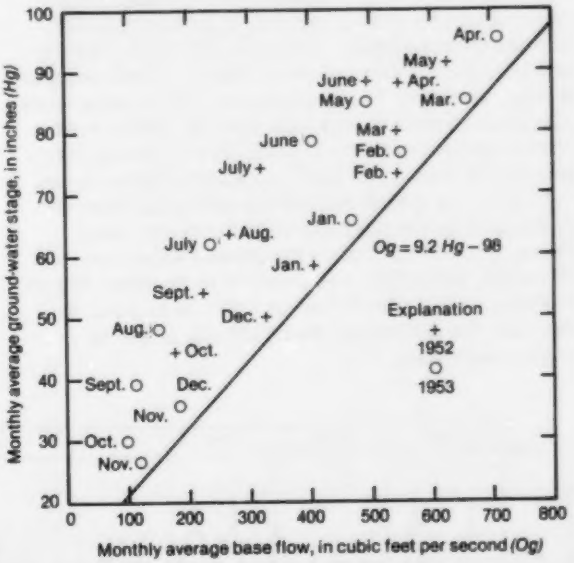


Figure 107. Relation of Monthly Average Base Flow to Ground-Water Stage in the Brandywine Creek Basin[22]

The rating curve in Figure 107 shows the relation between ground-water runoff and ground-water stage in the Brandywine Creek basin. Notice the elipitcal pattern of the data, which approach a straight line from October through March but then loops back during spring, summer, and early fall. Although confusing at first glance, the significance of the loop becomes readily apparent when the changes that occur in a ground-water reservoir throughout a 12-month period in a humid area are considered. From late fall to spring, the ground-water stage rises because there are little or no losses to evapotranspiration, soil moisture may be at or above field capacity, and ground-water recharge occurs. The water table reaches its peak during the spring runoff. From April to October, however, large quantities of ground water are removed by evapotranspiration, the soil moisture becomes so depleted there is little or no recharge, and the quantity of water in storage decreases because ground-water runoff exceeds recharge. Thus, the elipitcal shape of the data on the rating curve is controlled by evaportranspiration.

Using the rating curve, Olmsted and Hely[22] separated the Brandywine Creek hydrograph, shown in Figure 108, and found that over a six year period, ground-water runoff accounted for 67 percent of the total flow. This compares favorably with the 64 percent determined for North Branch Rancocas Creek in the coastal plain of New Jersey; 74 percent for Beaverdam Creek in the coastal plain of Maryland[21]; 42 percent for Perkiomen Creek, a flashy stream in the Triassic Lowland of Pennsylvania; and 44 percent for the Pomperaug River Basin, a small stream in Connecticut[23].

During certain times of the year, when the water table lies at a shallow depth and large quantities of water are lost by evapotranspiration, a single rating curve cannot be used to separate a hydrograph with any degree of accuracy. As Figure 109 shows, Schicht and Walton[24], in their study of Panther Creek basin in Illinois, developed two rating curves. One is used when evapotranspiration is very high and the other is used when evapotranspiration is small. Double rating curves also can be used to estimate evapotranspiration losses.

Evapotranspiration can also be calculated from the graph used by Olmsted and Hely in the case cited above. For example, when the ground-water stage was 80 inches, streamflow was expected to be about 550 cfs in February and March but only 400 cfs in June. In this case, the difference, about 150 cfs, is due to evapotranspiration.

Seepage or Dry-Weather Measurements

Seepage or dry-weather measurements consist of flow determinations made at several locations along a stream during a short time interval. It is essential that there be no surface runoff during these measurements. Many investigators prefer to conduct seepage runs during the stream's 90% flow duration, that is, when the flow is so low that is is equaled or exceeded 90 percent of the time.

It is often implied that the 90% flow duration is the only time the flow consists entirely of ground-water runoff. This is not necessarily the case. The 90% flow-duration period, depending on geographic location and climate, commonly occurs during the late summer and fall when soil moisture is depleted, there is little or no ground-water recharge, and the water table, having declined to its lowest level, has a low gradient. Under these conditions, ground-water runoff is minimal. However the physical aspect of the system changes following a recharge period and ground-water runoff may account for a substantial portion of the stream's flow. Hydrograph analyses, using techniques already described, may readily show that ground water provides 50 to 70 percent or more of the runoff. Therefore, the 90% flow may reflect only a small fraction of the total quantity of ground-water runoff.

Seepage measurements permit an evaluation of ground-water runoff (how much there is and where it originates) and provides clues to the geology of the basin as well. The flow of some streams increases substantially in a short distance. Under natural conditions this increase probably indicates deposits or zones of high permeability in or adjacent to the stream channel. These zones may consist of deposits of sand and gravel, fracture zones, solution openings in limestone, or merely by local facies changes that increase permeability. In gaining stretches, ground water may discharge through series of springs and seeps, along

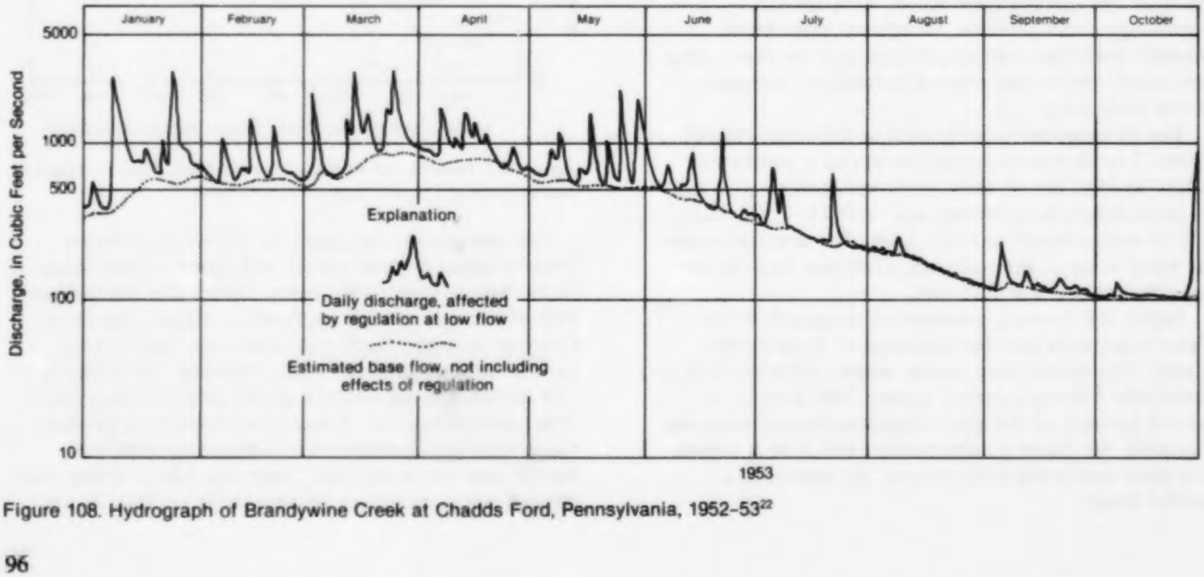


Figure 108. Hydrograph of Brandywine Creek at Chadds Ford, Pennsylvania, 1952-53[22]

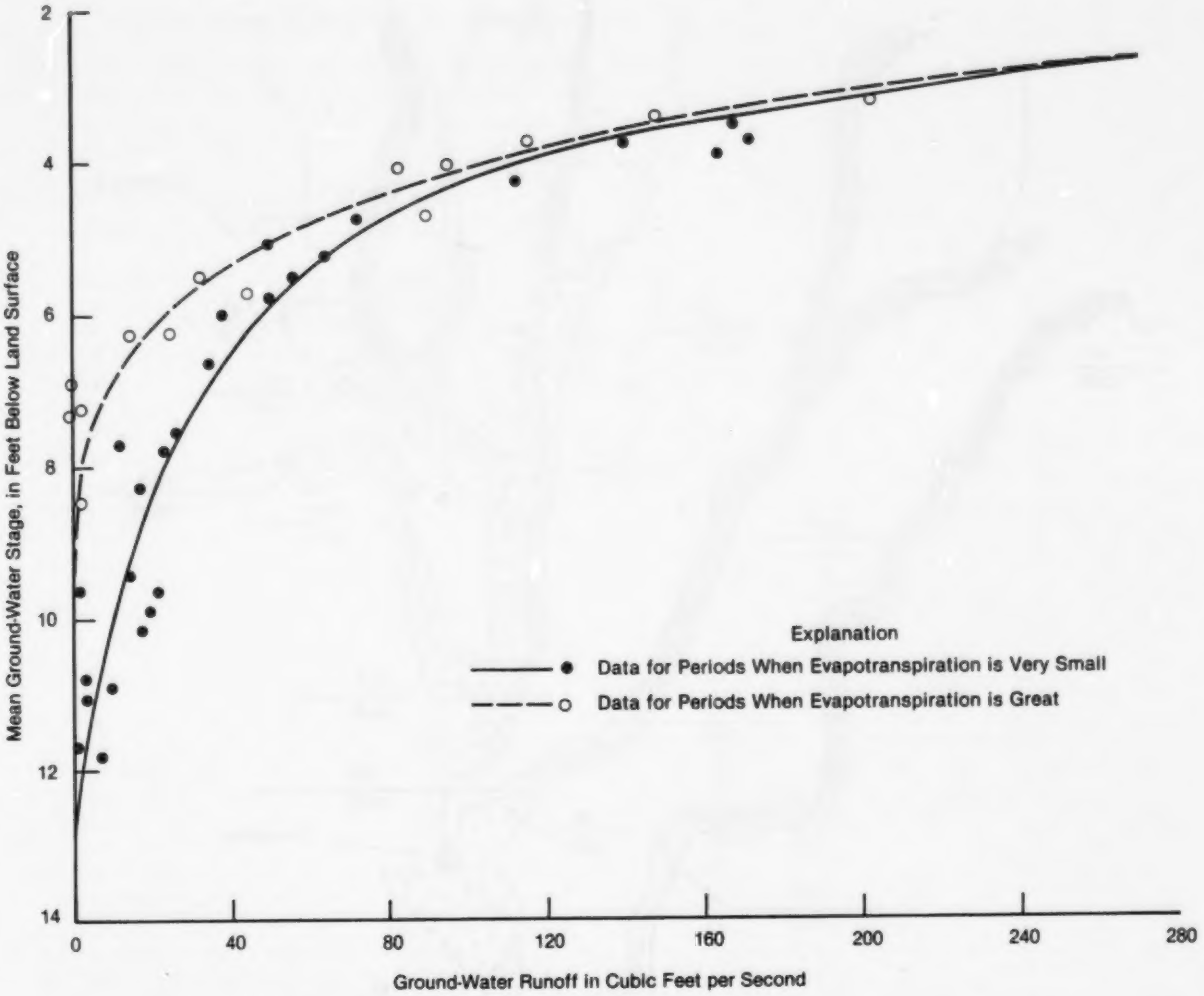


Figure 109. Rating Curves of Mean Ground-Water Stage Versus Ground-Water Runoff at Gaging Station 1, Panther Creek Basin[24]

valley walls or the stream channel, or seep upward directly into the stream. Areas of significant ground-water discharge may cause the formation of quicksand.

In areas where the geology and ground-water systems are not well known, stream flow data can provide a means of testing estimates of the ground-water system. If the streamflow data do not conform to the estimates, then the geology must be more closely examined.[25] For example, the northwest corner of Ohio is crossed by the Wabash and Fort Wayne moraines between which lies the St. Joseph River. As indicated by the Glacial Map of Ohio[26], the St. Joseph Basin consists mainly of till. However, low flow measurements show that the discharge of the river increases more than 14 cfs along its reach in Ohio, indicating that the basin contains a considerable amount of outwash. Thus, hydrologic studies indicate the need for geologic map modification.

On the other hand, geologic maps may indicate reasonable locations for constructing stream gaging stations for hydrologic monitoring networks. The main stem of the Auglaize River in northwestern Ohio rises from a mass of outwash that lies along the front of the Wabash moraine. The southwest-flowing river breaches the moraine near Wapakoneta and then flows generally north to its confluence with the Maumee River at Defiance. A gaging station is near Ft. Jennings in a till plain area and slightly above a reservoir on the Auglaize. In reality this gage measures, at a single point, the flow resulting as an end product of all causative hydrologic factors upbasin (ground-water runoff, surface runoff, slope, precipitation, use patterns, etc.)—it shows merely inflow into the reservoir. Low-flow measurements, however, indicate that nearly all of the baseflow is derived from outwash along the distal side of the Wabash moraine; there is no grain across the wide till-plain downstream. It would seem that the most logical stream

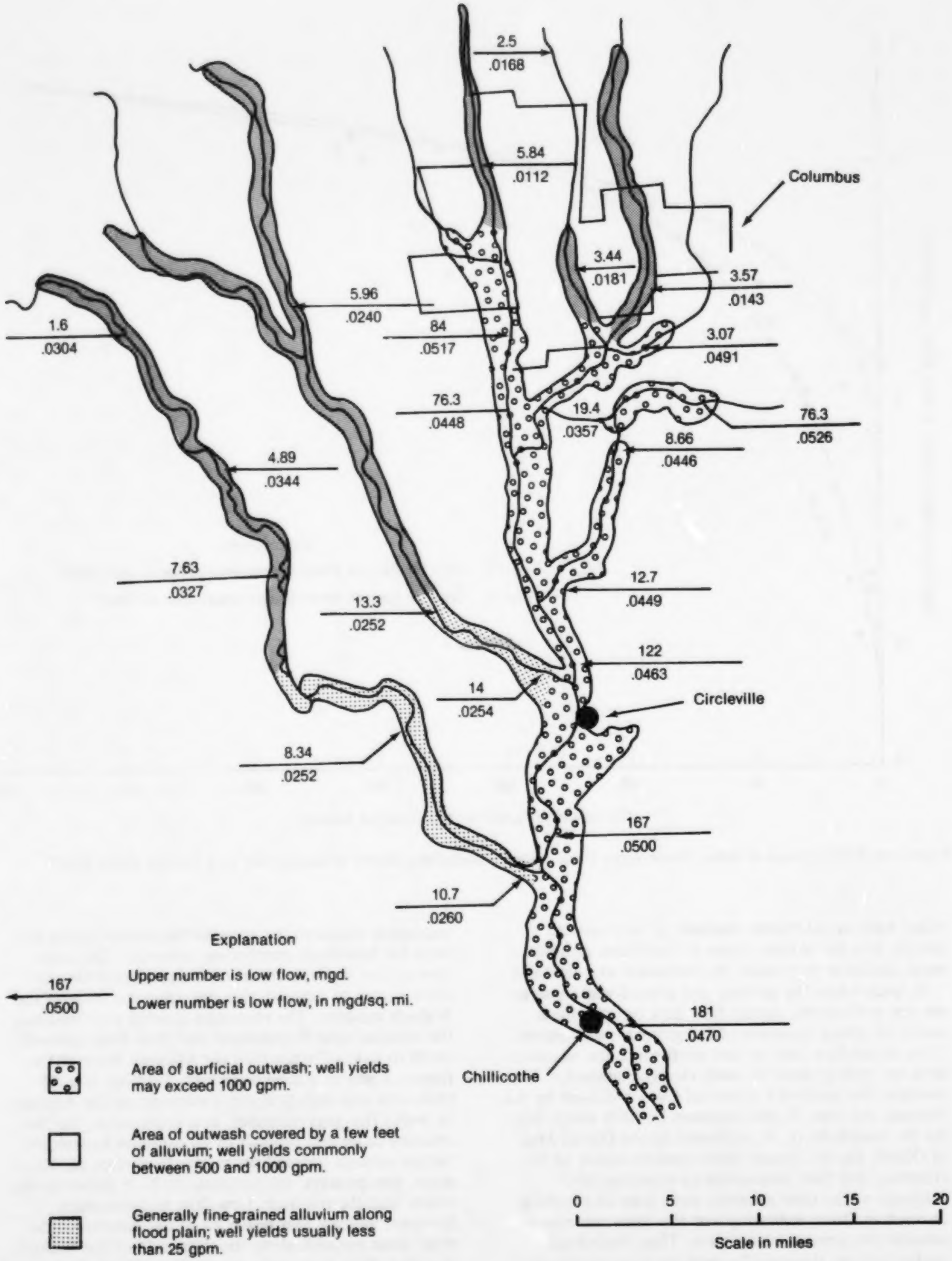

2.5
.0168
5.84
.0112
Columbus
3.44
.0181
3.57
.0143
5.96
.0240
1.6
.0304
84
.0517
3.07
.0491
76.3
.0448
19.4
.0357
76.3
.0526
8.66
.0446
4.89
.0344
7.63
.0327
13.3
.0252
12.7
.0449
122
.0463
14
.0254
Circleville
8.34
.0252
167
.0500
10.7
.0260
181
.0470
Chillicothe
Explanation
167
.0500
Upper number is low flow, mgd.
Lower number is low flow, in mgd/sq. mi.
Area of surficial outwash; well yields may exceed 1000 gpm.
Area of outwash covered by a few feet of alluvium; well yields commonly between 500 and 1000 gpm.
Generally fine-grained alluvium along flood plain; well yields usually less than 25 gpm.
0
5
10
15
20
Scale in miles

Figure 110. Discharge Measurements in the Scioto River Basin, Ohio

gage site for hydrologic evaluations would be at the breech in the Wabash moraine just downstream from the till-outwash contact.

Figure 110 shows a number of discharge measurements made in the Scioto River basin, which lies in a glaciated part of Central Ohio. The flow measurements themselves are important in that they show the actual discharge, in this case at about 90% flow. The discharge is reported as millions of gallons per day, instead of the usual cubic feet per second. The discharge at succeeding downstream sites on the Scioto River are greater than the sum of the flows immediately upstream. This shows that the river is gaining and that water is being added to it by ground-water runoff from the adjacent outwash deposits.

A paticularly useful method for evaluating streamflow consists of relating the discharge to the size of the drainage basin (cfs/sq.mi. or mgd/sq. mi. of drainage basin). As Figure 110 shows, this technique can be used to relate the flow index (cfs/sq.mi. or mgd/sq.mi.^{2}) to the geology and hydrology of the area. A cursory examination of the data shows that the flow indices can be conveniently separated into three distinctive units. These units are arbitrarily called Unit 1 (.01 to .020 mgd/mi.sq.), Unit 2 (.021 to .035 mgd/mi.sq.) and Unit 3 (.036 to .05 mgd/mi.sq.). The Olentangy River and Alum and Big Walnut Creeks fall into Unit 1, Big Darby and Deer Creeks into Unit 2, and the Scioto River, Walnut Creek, and the lower part of Big Walnut Creek into Unit 3. Notice that even though the latter watercourses fall into Unit 3, the actual discharge ranges widely—from 3.07 to 181 mgd.

Logs of wells drilled along the streams of Unit 1 show a preponderance of fine-grained material that contains only a few layers of sand and gravel and wells generally yield less than 3 gpm. Along Big Darby and Deer Creek, however, logs of wells and test holes indicate that several feet of sand and gravel underlie fine-grained alluvial material, the latter of which ranges in thickness from 5 to about 25 feet. Adequately designed and constructed wells that tap these outwash deposits produce as much as 500 gpm. Glacial outwash, much of it coarse grained, forms an extensive deposit through which the streams and rivers of Unit 3 flow. The outwash extends from the surface to depths that exceed 200 feet. Industrial wells constructed in these deposits, most of which rely on induced inflitration, can produce more than 1000 gpm. By combining the seepage data and well yields with a map showing the areal extent of the deposits characteristic of the stream valleys, the map in Figure 110, that indicates potential well yields in the area, was developed. The potential ground-water yield map relies heavily on streamflow measurement, but nonetheless, provides, with some geologic data, a good first cut approximation of ground-water availability.

Stream reaches characterized by significant increases in flow due to ground-water runoff, may also have unusual quality characteristics. In northern Ohio the discharge of a small stream, shown in Figure 111, that drains into Lake Erie increases over a 3-mile stretch from about 1 to more than 28 cfs and remains relatively constant thereafter. The increase begins at an area of springs where limestone, which has an abundance of solution openings, approaches land surface and actually crops out in the stream bottom. The till-limestone contact declines downstream eventually exceeding 90 feet in depth.

In the upper reaches of stream, baseflow is provided by ground water that discharges from the adjacent till. Since this water has been in the ground but a short time, the mineral content is low. Where streamflow begins to increase significantly, the limestone aquifer provides the largest increment. Furthermore, the bedrock water contains excessive concentrations of dissolved solids, hardness, and sulfate, and in this stretch calcite precipitates on rocks in the stream channel. The fish population in the upper reaches is exceedingly abundant until the stream reaches the limestone discharge zone. At this point, the population quickly diminishes and remains in a reduced state throughout the remaining length of the stream. No doubt the reduction in fish population is related to the quality of the water.

In describing the hydrology of Wolf Creek in east-central Iowa, Kunkle[19] used seepage measurements and water-quality data to determine the amount of ground-water runoff provided by alluvium and limestone. As Figure 112 shows, the 325-square mile basin is mantled by till and underlain by limestone and shale, but the valley itself contains about 40 feet of permeable alluvium. Well data show that the stream is hydraulically connected with the limestone aquifer along a 5-mile stretch and baseflow is provided by discharge from both the limestone and the alluvium. On either side of this reach the limestone potentiometric surface is below the stream bed.

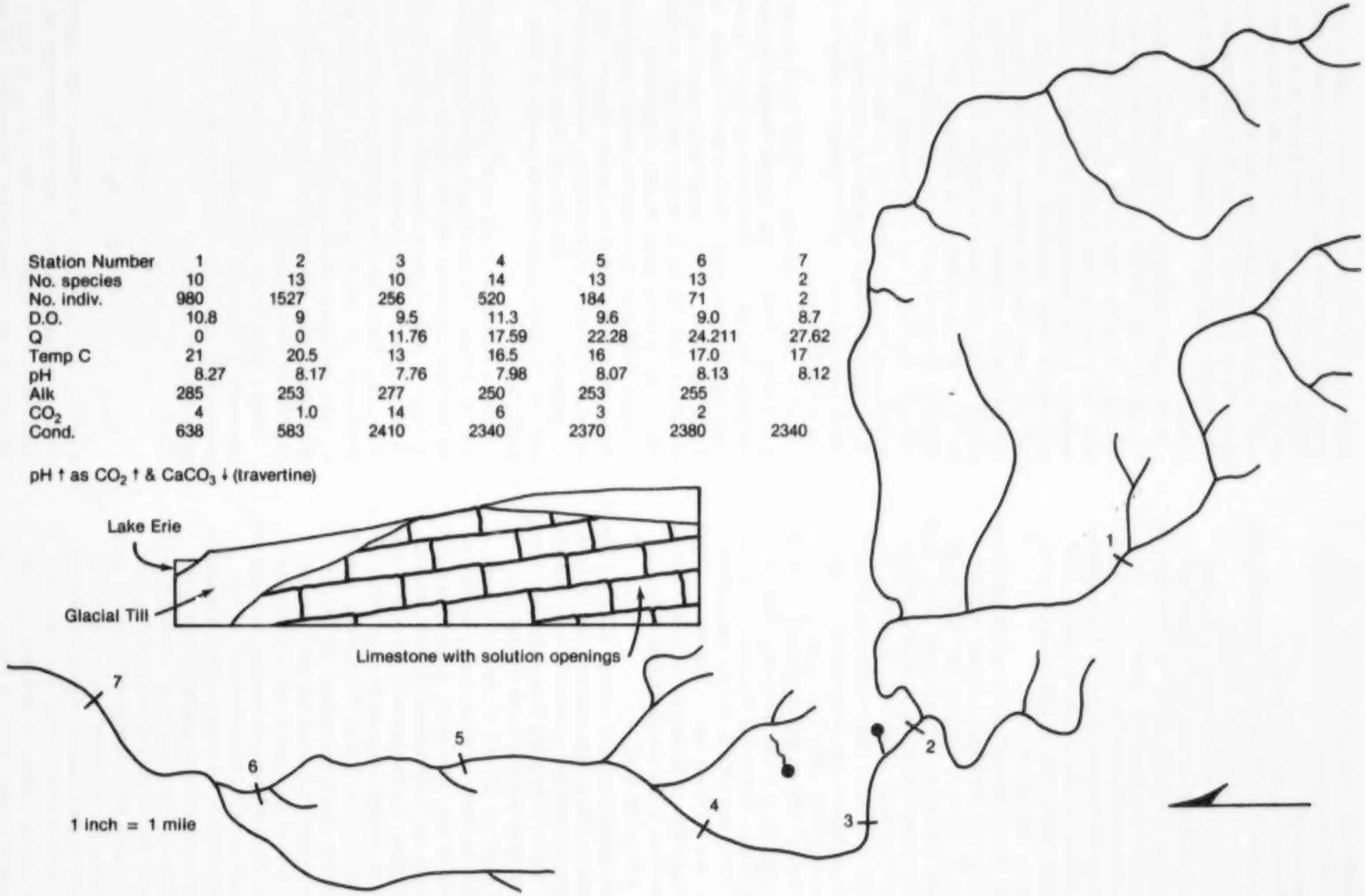
Measurements were made at three stations during low-flow conditions. The discharge 8 miles upstream from the limestone discharge area was 16.4 cfs (.086 cfs/sq.mi.), midway along the reach was 29.8 cfs (.0997 cfs/sq.mi.), and 7 miles downstream was 37.0 cfs (.114 cfs/sq.mi.). Water from the limestone has an average conductivity of 1330 micromhos, while that from the alluvium and upstream-derived baseflow average 475 micromhos. After mixing the surface water had a conductivity of 550 micromhos.

Using a slight modification of the equations given previously, it is possible to calculate the amount of ground-water runoff from the limestone in this reach under the given conditions.

$$C_i Q_i + C_a Q_a + C_b Q_b = C Q_o \tag{54}$$

$$Q_i + Q_a + Q_b = Q_o \tag{55}$$

where Q_i, Q_a, Q_b, Q_o are the discharge from upstream (inflow), from the alluvium, from the limestone, and from the outflow respectively, and C_i, C_a, C_b and C represent the conductivity of the inflow from upstream,



Station Number	1	2	3	4	5	6	7
No. species	10	13	10	14	13	13	2
No. indiv.	980	1527	256	520	184	71	2
D.O.	10.8	9	9.5	11.3	9.6	9.0	8.7
Q	0	0	11.76	17.59	22.28	24.211	27.62
Temp C	21	20.5	13	16.5	16	17.0	17
pH	8.27	8.17	7.76	7.98	8.07	8.13	8.12
Alk	285	253	277	250	253	255	
CO_2	4	1.0	14	6	3	2	
Cond.	638	583	2410	2340	2370	2380	2340

pH ↑ as CO_2 ↑ & $CaCO_3$ ↓ (travertine)

Figure 111. Green Creek Drainage Basin (Seneca and Sandusky Co.'s, Ohio)

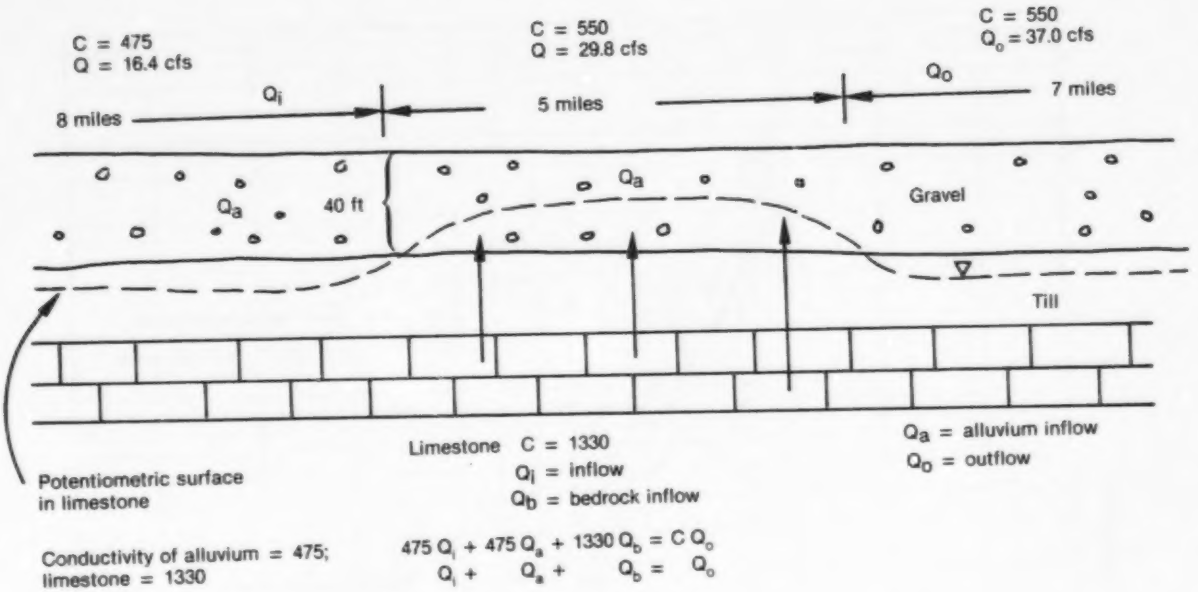


Figure 112. Generalized Hydrogeology of Wolf Creek, Iowa

from the alluvium, from the limestone, and from the outflow water. Substituting:

$$
\begin{aligned}
475\,Q_i + 475\,Q_a + 1330\,Q_b &= 550 \times 37 \\
-475\,Q_i - 475\,Q_a - 475\,Q_b &= 475 \times 37 \\
855\,Q_b &= 2775 \\
Q_b &= 3.2 \text{ cfs}
\end{aligned}
$$

Thus in this particular stretch, the limestone was providing about 3.2 cfs of the stream's total flow of 37 cfs.

Carrying the analyses a bit farther, we could assume that since the limestone provides, 3 to 4 cfs during baseflow, wells tapping the limestone in this stretch could provide a like amount without dewatering the system. Since 1 cfs = 450 gpm, wells could produce a total yield of 1350 to 1800 gpm. Using similar logic we could predict the minimum yield of wells tapping the alluvium, assuming that they would capture only the ground water runoff.

Temperature Surveys

The temperature of shallow ground water is nearly uniform, reflecting the mean annual air temperature of the region. The temperature of shallow ground water ranges from a low of about 37 degrees in the north-central part of the U.S. to more than 77 degrees in southern Florida. Of course at any particular site the temperature of ground water remains nearly constant. Surface water temperatures, however, range within wide extremes—freezing in the winter in northern regions and exceeding 100 degrees during hot summer days in the south. Mean monthly stream temperatures during July and August range from a low of 55 in the northwest to more then 85 degrees in the southeast.

During the summer where ground water provides a significant increment of flow, the temperature of surface water in a gaining reach will decline. Conversely, during winter the ground water will be warmer than that on the surface and although ice will normally form, parts of a stream may remain open because of the inflow of the warmer ground water. In central Iowa, for example, winter temperatures commonly drop below zero and ice quickly forms on streams, ponds, and lakes. The ground-water temperature in this region, however, is about 52 degrees and, if a sufficient amount is discharging into a surface-water body, the temperature may remain above 32 degrees and the water will not freeze. In the summer, the relatively cold ground water (52 degrees) mixes with the warm surface water (more than 79 degrees) producing a mixture of water colder than that in non-gaining reaches.

Field examination of winter aerial photography may show places where ice is either absent or thin. In the summer it is possible to float down a river, periodically measuring the temperature. Ground-water discharge areas are detected by temperature decrease. A third method of detection is by means of an aircraft-mounted thermal scanner. This sophisticated instrument is able to detect slight differences in temperature and would probably be more accurate than thermometry or low altitude aerial photography.

Flow-Duration Curves

A flow-duration curve shows the frequency of occurrence of various rates of flow. It is a cumulative frequency curve prepared by arranging all discharges of record in order of magnitude and subdividing them according to the percentages of time during which

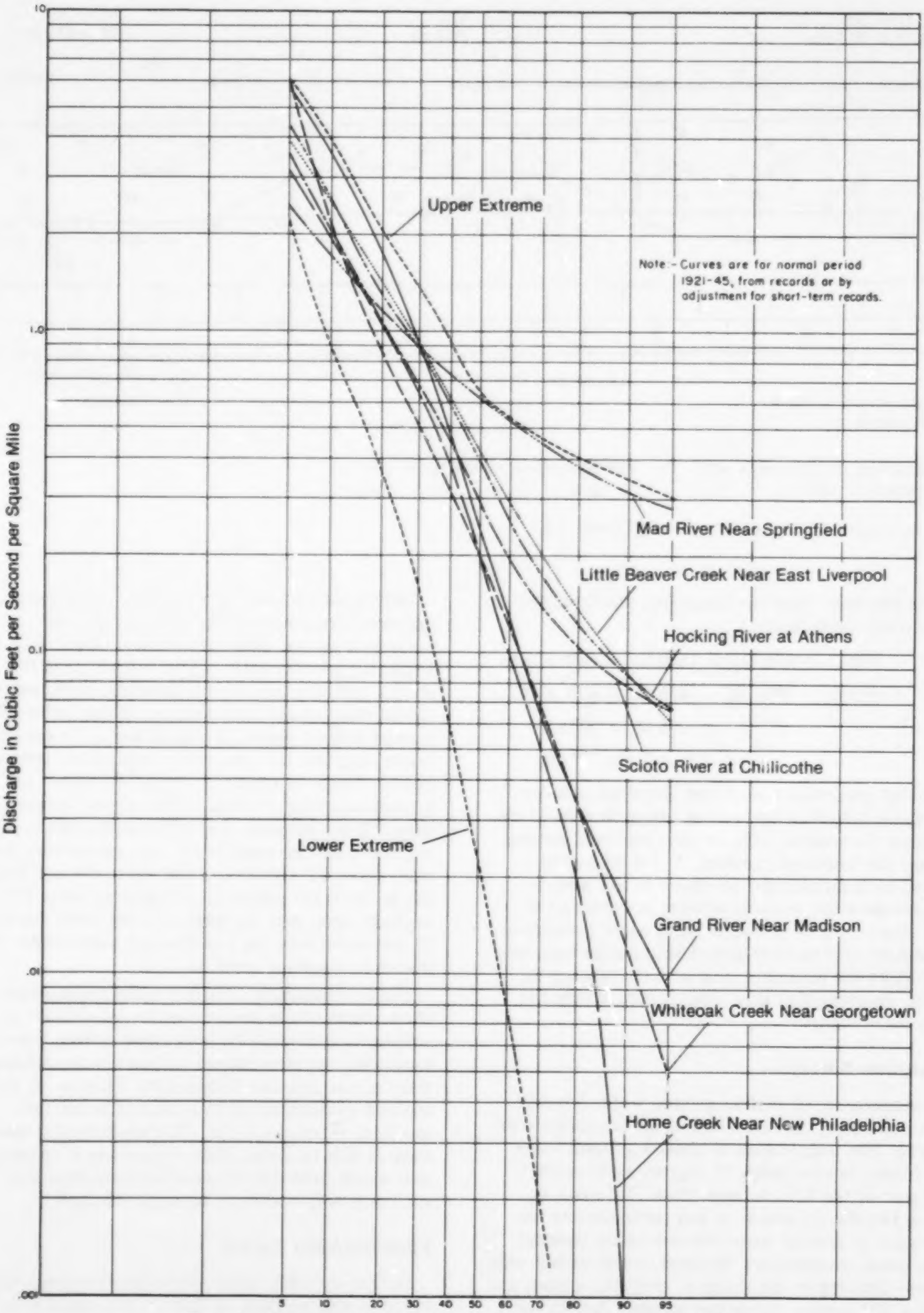


Figure 113. Flow-Duration Curves for Selected Ohio Streams.[27]

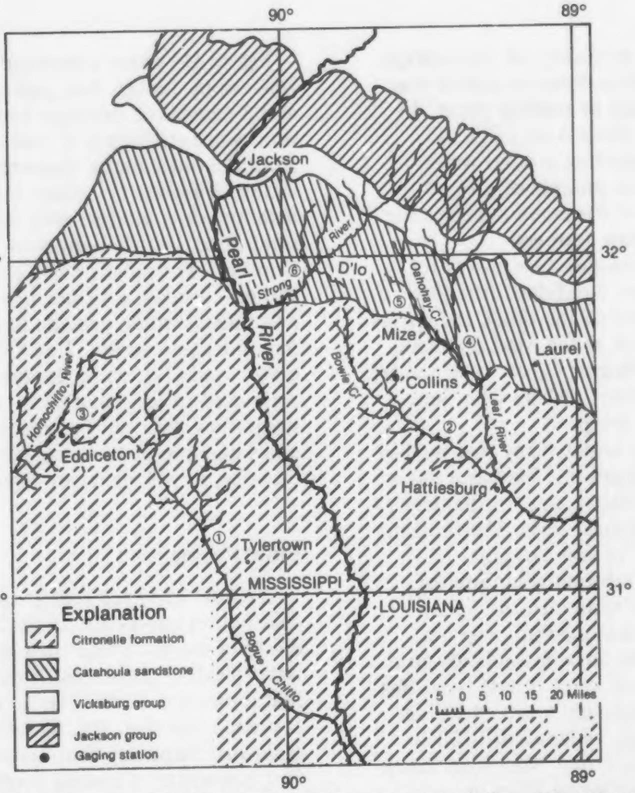


Figure 114. Geologic Map of Area in Southern Mississippi Having Approximately Uniform Climate and Altitude

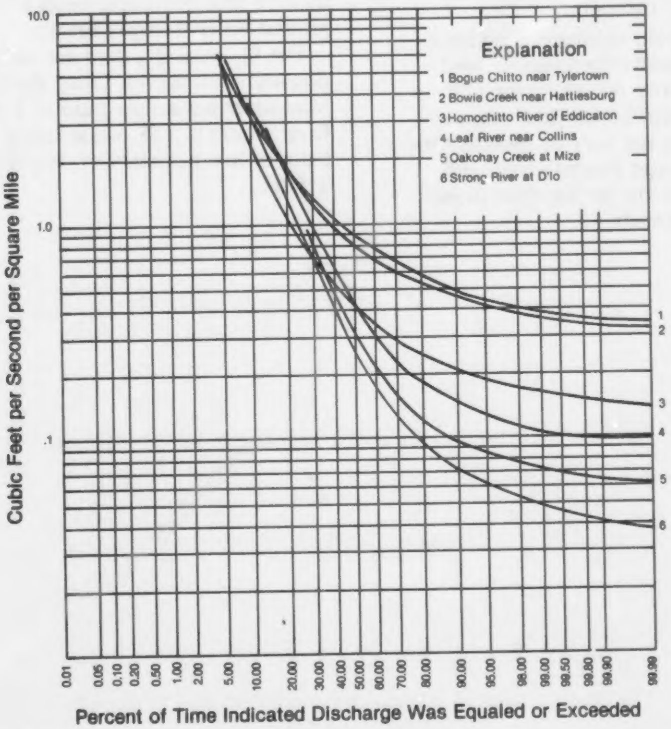


Figure 115. Flow-Duration Curves for Selected Mississippi Streams, 1939–48

specific flows are equaled or exceeded; all chronologic order or sequence is lost.[27] Flow-duration curves may be plotted on either probability or semilog paper. In either case, the shape of the curve is an index of natural storage in a basin, including ground water. Since dry-weather flow consists entirely of ground-water runoff, the lower end of the curve indicates the general characteristics of shallow aquifers.

Figure 113 shows several flow-duration curves for Ohio streams. During low-flow conditions (the flow equaled or exceeded 90 percent of the time), the curves for several of the streams, such as the Mad, Hocking, and Scioto River, and Little Beaver Creek trend toward the horizontal, while Grand River, White Oak, and Home Creeks all remain very steep.

Mad River flows through a broad valley that is filled with very permeable sand and gravel. The basin has a large ground-water storage capacity and, consequently, the river maintains a high sustained flow. The Hocking river locally contains outwash in and along its floodplain, which provides a substantial amount of ground-water runoff. Above Columbus, the Scioto River crosses thin layers of limestone that crop out along the stream valley and the adjacent uplands are covered with glacial till; ground-water runoff from this reach is relatively small. Immediately south of Columbus, however, the Scioto Valley widens and is filled with coarse outwash. The reason that Mad River has a higher low-flow index than the Scioto River at Chillicothe is because the Mad River receives ground-water runoff throughout its entire length, while the flow of the Scioto River increases significantly only in the area of outwash south of Columbus.

White Oak and Home Creeks originate in bedrock areas where relatively thin alternating layers of sandstone, shale, and limestone crop out along the hill sides. The greater relief in these basins promotes surface runoff and the rocks are not very permeable. Obviously the ground-water storage characteristics and potential yield of these basins are far less than those filled or partly filled with outwash.

Figure 114 shows a geologic map of a part of southern Mississippi. Notice that gaging stations at Sites 1, 2, and 3 record the drainage from the Citronelle Formation, while stations 4, 5, and 6 represent the drainage from the older rocks. Respective flow-duration curves, shown in Figure 115, show that stations 1 and 2 have high low-flow indices, with station 3 a relatively close third. The high flow-duration indices indicate that the Citronelle Formation has a greater ground-water storage capacity, a higher rate of natural recharge, and presumably would provide larger yields to wells than the underlying strata. This formation consists of sand, gravel, and clay, while the other strata are generally composed of finer materials. Thus it would appear that streamflow data can be used as an aid to a better understanding of the permeability and infiltration capacity, as well as facies changes, of geologic units.

Flow Ratios

Walton[28] reported that grain-size frequency-distribution curves are somewhat analogous to flow-duration curves in that their shapes are indicative of water-yielding properties of deposits. He pointed out that a measure of the degree to which all of the grains approach one size, and therefore, the slope of the grain-size frequency distribution curve, is the sorting. One parameter of sorting is obtained by the ratio $(D_{25}/D_{75})^{1/2}$. Walton modified this equation by replacing the 25 and 75 percent grain-size diameters with the 25 and 75 percent flow. In this case a low ratio is indicative of a permeable basin or one that has a large ground-water storage capacity.

The Q_{25} and Q_{75} data are easily obtainable from flow-duration curves. Using the data from Figure 113, Mad River has a flow ratio of 1.58 and the Scioto River's ratio is 2.58, while Home Creek, typifying a basin of low permeability, has the highest ratio which is 5.16.

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Chapter 5

Ground-Water Pollution

Long-Term Effects

For millennia, man has disposed of his waste products in a variety of ways. The disposal method might reflect convenience, expedience, expense, or best available technology, but in many instances, leachate from these wastes have come back to haunt later generations. This is largely because we have not thought out the consequences of our actions. Ground-water pollution may lead to problems of inconvenience, such as taste, odor, color, hardness, or foaming; but the pollution problems are far more serious when pathogenic organisms, flammable or explosive substances, or toxic chemicals or their by-products are present, particularly when long-term health effects are unknown.

Individual polluted ground-water sites generally are not large, but once polluted, ground water may remain in an unusable or even hazardous condition for decades or even centuries. The typically low velocity of ground water prevents a great deal of mixing and dilution; consequently, a contaminant plume may maintain a high concentration as it slowly moves from points of recharge to zones of discharge.

An oil-field brine holding pond was constructed adjacent to a producing well in central Ohio in 1968. Two years later when the well was plugged, the holding pond was filled, graded, and seeded. The chloride concentration in the ground water in the vicinity of the former pond still exceeded 36,000 mg/l some 10 years after the operation began and 8 years after reclamation.

Scores of brine holding ponds were constructed in central Ohio during an oil boom in 1964; many are still in use. In 1978 a number of test holes were constructed within 200 feet of one such pond. Within this area shallow ground water contained as much as 50,000 mg/l of chloride. Moreover, brine-contaminated ground water provides part of the flow of many streams and this has caused degradation of surface-water quality.[29, 30, 31]

Documentation of the migration of leachate plumes originating at garbage dumps and landfills is becoming increasingly abundant. Data show that under certain hydrologic conditions leachate plumes can move considerable distances and degrade ground water throughout wide areas. Furthermore, the problem is worldwide. Exler[32] described a situation in southern Bavaria, Germany, where a landfill has been in operation since 1954. The wastes are dumped into a dry gravel pit. As Figure 116 illustrates, data collected from 1967 to 1970 showed the narrow lense-shaped plume had migrated nearly 2 miles.

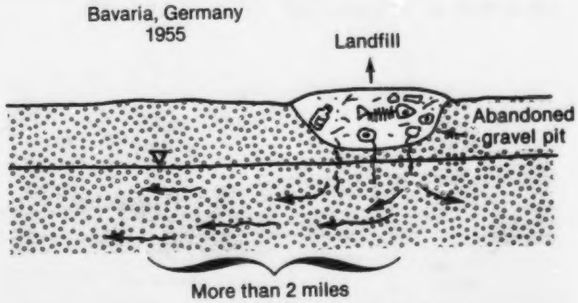


Figure 116. Leachate from a Landfill in Bavaria Has Migrated More Than 2 Miles and the Ground Water Has Been Degraded for Nearly 25 Years.

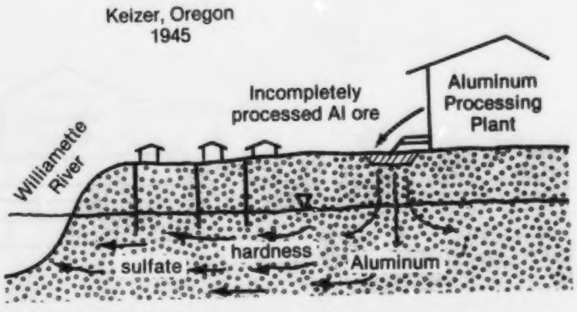


Figure 117. Thirty-Three Years After Disposal Began the Leachate from Aluminum Ore and Mill Tailings is Still a Problem in Keizer, Oregon.

As Figure 117 illustrates, incompletely processed aluminum ore was dumped into a borrow pit in Keizer, Oregon from August 1945 to July 1946.[33] The ore and mill tailings had been treated with sulfuric acid and ammonium hydroxide. When first recognized by local residents in 1946, the ground water was contaminated by more than 1,000 mg/l of sulfate; many shallow domestic wells tapping the Recent alluvium were contaminated. In the Spring of 1948 the waste was removed from the borrow pit. Two wells, reportedly capable of producing more than 700 gpm (gallons per minute) were installed near the pit and the contaminated groundwater was pumped to waste for several months. By 1964 the contaminants had migrated more than a mile. No doubt some of the contaminants are still in the ground water at Keizer.

A well-documented study by Perlmutter and others[34] showed that disposal of chromium and cadmium-rich plating wastes from an aircraft plant on Long Island during a 20-year period contaminated a shallow aquifer. Figure 118 illustrates this study. The contamination was first discovered in 1942, and by 1972 the degraded ground water zone was about 4,200 feet long and 1,000 feet wide. The 1972 study demonstrated that the chromium-cadmium enriched cigar-shaped plume "had not only reached Massapequa Creek but was present in the stream as well as in the beds beneath it."[35]

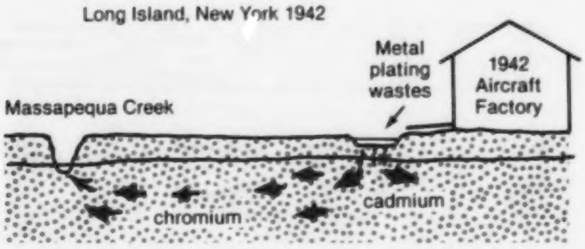


Figure 118. More Than 36 Years After Disposal of Plating Wastes Began, the Ground Water Remains Polluted in South Farmingdale.

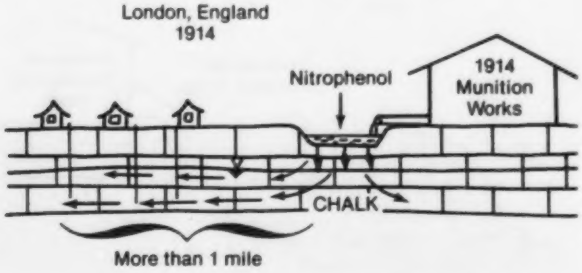


Figure 119. The Picric Acid, Which Has Been Found in the Ground Water Near London for Decades, Originated at a World War I Munitions Plant.

During the middle and late 1930's grasshopper infestations were stripping the vegetation throughout wide areas in the Northern Great Plains. In western Minnesota partial control was obtained by a grasshopper bait consisting of arsenic, bran, and sawdust. Eventually the leftover bait was buried. In May 1972, a contractor drilled a well near his office and warehouse on the outskirts of a small town. During the next two and a half months 11 of the 13 individuals employed at the site became ill; two were hospitalized. They were suffering from arsenic poisoning. One sample of water from the well contained 21 mg/l of arsenic. Analyses of soil from the site revealed arsenic concentrations ranging from 3,000 to 12,000 mg/l. Apparently the well was drilled in the vicinity of the grasshopper bait disposal site, which had long been forgotten by the local residents.[36]

Wastes from munitions works include picric acid, a toxic, intensely bitter, pale yellow substance. Picric acid is not readily removed by traditional water treatment methods and its migration through either the unsaturated zone or the saturated zone does not appear to neutralize it.

During the World War I years of 1914–1918, wastes from the manufacture of explosives at a plant near the Thames River just northeast of London, England, were placed in abandoned chalk pits. Figure 119 illustrates the migration of these wastes. In the early 1920's water from a nearby well was first reported to have a yellow tint.[37] Additional water samples collected between 1939 and 1955 also contained a characteristic yellow picric acid tint. Sampling ceased in 1955 when the pump was removed. By 1942 the pollutants had migrated at least a mile as indicated by another contaminated well. There is no reason to believe that the picric acid has been flushed from the aquifer. The ground water has certainly been polluted for 40 years, quite probably for more than 70 years, and very likely will be polluted for many more years to come.

Because of high evaporation and low recharge, waste disposal in arid regions can lead to long-lived groundwater quality problems. In the first place, salts are concentrated by evaporation to form highly mineralized fluids. Secondly, water supplies may not be readily available and, therefore, every effort must be made to protect existing sources.

Ground-water contamination in the desert environment near Barstow, California, was described by Hughes.[38] Beginning around 1910, waste fuel oil and solvents from a railroad system were discharged to the dry floor of the Mojave River near Barstow. The first municipal sewage treatment plant was constructed in 1938; the effluent was discharged to the riverbed. Sewage treatment facilities were enlarged in 1953 and 1968. Effluent disposal was dependent on evaporation and direct percolation into the alluvial deposits.

At the U.S. Marine Corps base near Barstow, industrial and domestic waste treatment facilities first became operational in 1942; effluent disposal relied on direct percolation and evaporation. Some of the effluent was used to irrigate a golf course. Other sources of groundwater contamination were two nearby mining and milling operations.

As Figure 120 shows, analysis of well waters collected during the Spring of 1972 indicated the existence of two zones of contaminated ground water in the alluvial deposits of the Mojave River. The deeper zone, originating from the 1910 disposal area, exceeded 1,800 feet in width and extended nearly 4½ miles in a downgradient direction. Its upper surface lies 60 or more feet below land surface. The second or shallow zone originates at the sewage treatment lagoon installed in 1938 and at the Marine Corps golf course. This zone consists of two apparently separate plumes. The upgradient plume extends nearly 2 miles downstream, while the plume originating at the golf course is nearly a mile long; the plumes are about 700 feet wide. Hughes estimated that the pollution fronts are moving at a rate of 1 to 1.5 feet per day. The Marine Corps well field lies in the path of these plumes; several domestic wells have already been contaminated. In this instance poor waste disposal practices, beginning nearly 75 years ago may cause water-supply problems at the Marine Corps base unless expensive corrective measures are undertaken.

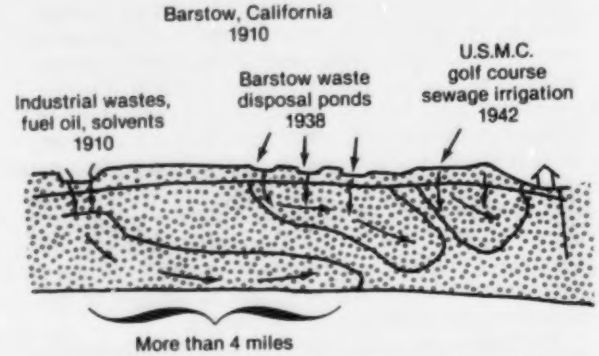

Barstow, California
1910
Industrial wastes, fuel oil, solvents 1910
Barstow waste disposal ponds 1938
U.S.M.C. golf course sewage irrigation 1942
More than 4 miles

Figure 120. Waste Disposal Beginning Nearly 70 Years Ago at Barstow, California is Now Threatening an Important Well Field at the Nearby Marine Base.

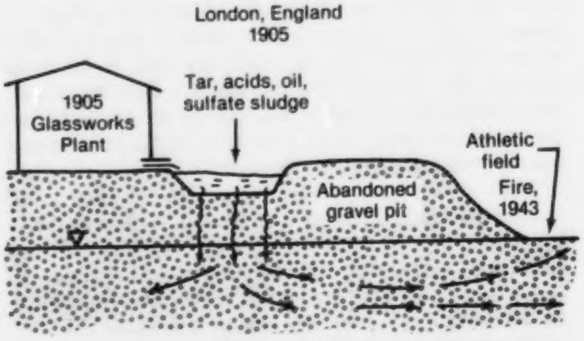


Figure 121. Ground-Water Pollution by Wastes from a Gasworks Plant Near London Has Even Created a Fire Hazard.

From 1905 to 1967 wastes from a gasworks plant were deposited in abandoned gravel pits along the Lee River near Waltham Cross, a few miles northwest of London, England.[39] Figure 121 shows that the tar acids, oils, and sulfate sludge infiltrated to contaminate the ground water over a wide area. Apparently the pollution was first detected in 1935, some 30 years after disposal began. At that time oil, floating on the ground water, emerged at land surface. Continual but slow accumulation of oil on and near the land surface led to hazardous conditions and, in 1943, the oil was ignited. Contaminated ground water was also encountered in new excavations where it appeared as high concentrations of sulfate in 1958 and as oily waters in 1961. In 1965, oily liquids also seeped into Pymmes Brook and the Lee River Navigation Channel following a substantial rise in the water table after heavy rains. Additional surface-water degradation occured in 1966 because of the discharge of oil from streamside seepage zones.

Ground water in the surficial sand and gravel deposit was contaminated over a wide area. Fortunately, most water supplies in this region are pumped from an underlying chalk, which generally is separated from the gravel by the London Clay. It is evident from this example that waste disposal, which began 80 years ago, continues to be troublesome and that ground-water contamination can indeed become a fire hazard.

All ground-water pollution is not necessarily bad. Inhabitants of Crosby, a small village in northwestern North Dakota, believed they produced the best coffee in the State because the water from which it was made contained "body". The rather highly mineralized water (dissolved solids = 2,176 mg/l, sulfate = 846 mg/l, chloride = 164 mg/l, and nitrate = 150 mg/l) used for brewing the coffee was obtained exclusively from an old dug well. The well, however, was constructed, probably near the turn of the century, at the site of the local river livery stable. Livestock wastes provided the peculiar flavor so characteristic of the coffee made in Crosby.[40]

The manufacture of soda ash, caustic soda, chlorine, and allied chemicals began at Barberton, Ohio, shortly before the turn of the century. The plant discharged a mixture of calcium and sodium chlorides directly to the Tuscarawas River and to retention ponds. The discharge of chloride in 1966 averaged 1,500 tons per day.[41] These wastes have led to serious ground-water pollution problems in eastern Ohio and have necessitated abandonment of streamside well fields at Barberton in 1926 and at Massillon and Coshocton in 1953.

Municipal wells at Zanesville, more than 135 river miles downstream from Barberton, have also been adversely affected by the chloride induced into the watercourse aquifer from the contaminated Muskingum River. Due to high treatment costs Zanesville officials considered abandoning their well field in 1963. At the confluence of the Muskingum and Ohio Rivers, about 220 river miles below Barberton, is the city of Marietta. Almost 30 years ago, Marietta officials were concerned over the marked increase in chloride in municipal wells during the preceding 10 years.[42] The cause, of course, was induced infiltration of the chloride-rich Muskingum River water.[29]

It is evident that decades of poor waste-disposal practices at Barberton seriously impaired streamside aquifers and well fields for a distance of over 200 river miles. The soda ash plant at Barberton was closed in 1973 and waste discharges substantially reduced. Presumably, these water-quality problems will decrease in severity over the next several years, after a history of 90 years or more.

According to Mink and others[43] mining operations in the Coeur d'Alene district of northern Idaho have been continuous for more than 90 years. Unfortunately, leaching of the ancient mining and milling wastes is now affecting the chemical quality of ground water in several areas, including Canyon Creek basin near Wallace. Here high concentrations of zinc, lead, copper, and cadmium occur in both ground water and soil samples.

In 1884 striking miners set fire to several deep coal mines in the vicinity of New Straitsville, Ohio. Still burning uncontrollably, the fires were started by disgruntled workers who rolled burning wood-filled coal cars into the shafts that honeycomb the ground under the town. In the years since, many wells have become contaminated, dried up, or produce water hot enough to make instant coffee.

Disposal of domestic, industrial, and municipal wastes, which probably began around 1872 through wells and sinkholes tapping a permeable limestone aquifer, was the birth of a contaminated area that now encloses some 75 square miles. By 1919 the practice of disposing of sewage at the northern Ohio town of Bellevue was well established and many wells had been contaminated. In the early 1960's some wells were reported to yield easily recognizable raw sewage.[44] This problem began more than a hundred years ago and remains to this day.

A gasworks plant was built at Norwich, England, in 1815 and abandoned in 1830. Phenolic compounds, originating from whale oil, infiltrated and remained in the underlying chalk for at least 135 years when it contaminated a newly drilled well in 1950.[40, 45] These organic compounds, no doubt, are still there 170 or so years later.

Sources of Ground-Water Contamination

As water moves through the hydrologic cycle, its quality changes in response to differences in the physical, chemical, and biological environments through which it passes. The changes may be either natural or man-influenced; in some cases they can be controlled, in other cases they cannot, but in most cases they can be managed in order to limit adverse water-quality changes.

The physical, chemical, and biological quality of water may range within wide limits even though there are no man-made influences. In fact, it is often impossible or at least difficult to distinguish the origin (man-made or natural) of many water-quality problems. The natural quality reflects the types and amounts of soluble and insoluble substances with which the water comes in contact. Surface water generally contains less dissolved solids than ground water, although at certain times (generally during low flow rates) in areas where ground-water runoff is the major source of streamflow, the quality of both surface water and ground water is similar. During periods of surface runoff, streams may contain large quantities of suspended materials and, under some circumstances, a large amount of dissolved solids. Most commonly, however, during high rates of flow the water has a lower dissolved-mineral concentration.

Although the chemical quality of water in surficial or shallow aquifers may range within fairly wide limits from one time to the next, deeper ground water is characterized by nearly constant chemical and physical properties, at least on a local scale where the aquifer is unstressed by pumping. As a general rule, the dissolved-solids content increases with depth and with the time and distance the water has traveled in the ground. A few uncommon water-quality situations exist throughout the country, reflecting unusual geologic and hydrologic conditions. These include, among others, thermal areas and regions characterized by high concentrations of certain elements, some of which may be health hazards.

For centuries man has been disposing of his waste products by burning, placing them in streams, storing them on the ground, or putting them in the ground using various methods. Man-made influences on stream-water quality reflect not only waste discharge directly into the stream, but also include highly mineralized or polluted surface runoff, which can carry a wide variety of substances. Another major influence on surface-water quality is related to the discharge of ground water into the stream. If the adjacent ground water is polluted, stream quality tends to deteriorate. Fortunately in the latter case, the effect in the stream generally will not be as severe as it is in the ground, due to dilution of the pollutant. See Reference 31 for example.

The quality of ground water is most commonly affected by waste disposal. One major source of pollution is the storage of waste materials in excavations, such as pits or mines. Water-soluble substances that are dumped, spilled, spread, or stored on the land surface or in excavations may eventually infiltrate to pollute ground-water resources. Ground water is also polluted by the disposal of fluids through wells and, in limestone terrains, through sinkholes directly into aquifers. Likewise, infiltration of highly mineralized surface water has been a major cause of underground pollution in several places. Irrigation tends to increase the mineral content of both surface and ground water. The degree of severity of pollution in cases such as these is related to the hydrologic properties of the aquifers, the type and amount of waste, disposal techiniques, and climate.

A major and widespread cause of ground-water quality deterioration is pumping, which may cause the migration of more highly mineralized water from surrounding strata to the well. The migration is directly related to differences in hydrostatic head between adjacent water-bearing zones and to the hydraulic conductivity of the strata. In coastal areas pumping may cause sea water to invade a fresh water aquifer. In parts of coastal west Florida, wild-flowing, abandoned artesian wells have salted, and consequently ruined, large areas of formerly fresh or slightly brackish aquifers.

The list in Table 10 shows that man-influenced ground-water quality problems are most commonly related to: (1) water-soluble products that are placed on the land surface and in streams; (2) substances that are deposited or stored in the ground above the water table; and (3) material that is stored, disposed of, or extracted from below the water table. Many of the pollution problems related to these situations are highly complex, and some are not well understood.

Table 10. Sources of Ground-Water Quality Degradation

Ground-Water Quality Problems that Originate on the Land Surface

1. Infiltration of polluted surface water
2. Land disposal of either solid or liquid wastes
3. Stockpiles
4. Dumps
5. Disposal of sewage and water-treatment plant sludge
6. De-icing salt usage and storage
7. Animal feedlots
8. Fertilizers and pesticides
9. Accidental spills
10. Particulate matter from airborne sources

Ground-Water Quality Problems that Originate in the Ground Above the Water Table

1. Septic tanks, cesspools, and privies
2. Holding ponds and lagoons
3. Sanitary landfills
4. Waste disposal in excavations
5. Leakage from underground storage tanks
6. Leakage from underground pipelines
7. Artificial recharge
8. Sumps and dry wells
9. Graveyards

Ground-Water Quality Problems that Originate in the Ground Below the Water Table

1. Waste disposal in well excavations
2. Drainage wells and canals
3. Well disposal of wastes
4. Underground storage
5. Secondary recovery
6. Mines
7. Exploratory wells
8. Abandoned wells
9. Water-supply wells
10. Ground-water development

Ground-Water Quality Problems that Originate on the Land Surface

Infiltration of Polluted Surface Water. The yield of many wells tapping streamside aquifers is sustained by infiltration of surface water. In fact, Figure 122 shows that more than half of the well yield may be derived directly from induced recharge from a nearby stream, which may be polluted. As the induced water migrates through the ground, a few substances are diluted or removed by filtration and sorption. This is especially true where the water flows through filtering materials, such as sand and gravel, particularly if these materials contain some soil organic matter. Filtration is less likely to occur if the water flows through large openings, such as those that occur in carbonate aquifers. Many pollutants, for example chloride, nitrate, and many organic compounds, are highly mobile, move freely with the water, and are not removed by filtration.

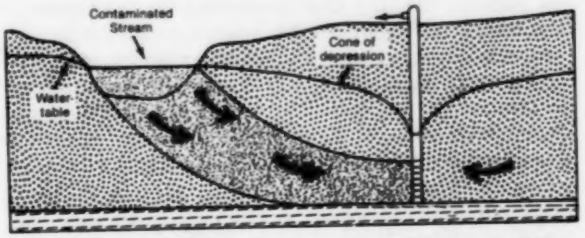


Figure 122. Induced Infiltration of Contaminated Stream Water May Lead to Ground-Water Pollution.

Examples of ground-water supplies being degraded by induced recharge of polluted surface water are both numerous and widespread. In the greatest number of cases, the pollution originated from the disposal of municipal or industrial waste directly into the stream, which was induced by pumping into adjacent aquifers. In hydrologic situations such as these, months or perhaps years may be required for the polluant to advance from the stream into the well. Once at the well, however, the aquifer between the well and the stream may be completely polluted, requiring years to recover once the source has been eliminated.

Land Disposal of Either Solid or Liquid Waste Materials. One of the major causes of ground-water pollution is the disposal of waste materials directly onto the land surface. Examples include manure, sludges, garbage, and industrial wastes. The waste may occur as individual mounds or it may be spread out over the land. If the waste material contains soluble products, they will infiltrate the land and may lead to ground-water pollution. Similar problems occur in the vicinity of various types of stockpiles.

Stockpiles. Perhaps the prime example of ground-water pollution caused by stockpiles is storage of de-icing salt (sodium and calcium chloride) used for highway snow and ice control. Not uncommonly, tons of salt are sim-

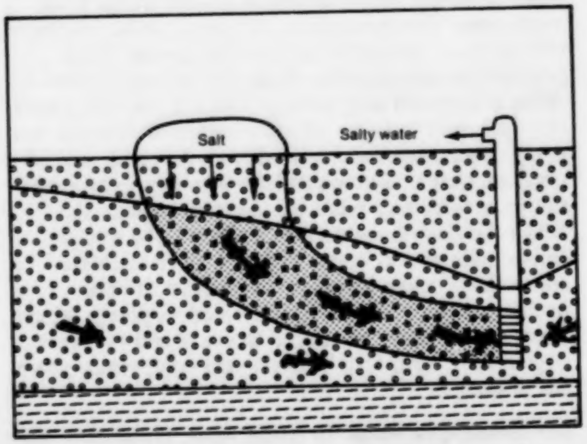


Figure 123. Leaching of Solids at the Land Surface. The Possibility of Ground-Water Pollution Under These Conditions is Rarely Anticipated.

ply piled on the land surface awaiting use. As Figure 123 shows, the highly soluble material rapidly dissolves and either infiltrates or runs off into streams. In recent years, many highway officials have provided some protection for salt stockpiles by covering them with plastic sheets or storing the salt in sheds. This is not necessarily done to protect adjacent water resources, but merely to preserve the salt.

Dumps. In the past few years, investigators have begun to take a serious look at the environmental effects of dumps. As rainwater infiltrates through trash in a dump, it accumulates a wide variety of chemical and biological substances. The resulting fluid, or leachate, may be highly mineralized. As the leachate infiltrates, some of the substances it contains are removed or degraded. As Figure 124 shows, eventually the leachate may reach the water table where it flows in the direction of the regional ground-water gradient or toward a well. (In some places "sanitary fills" or dumps lie below the water table.)

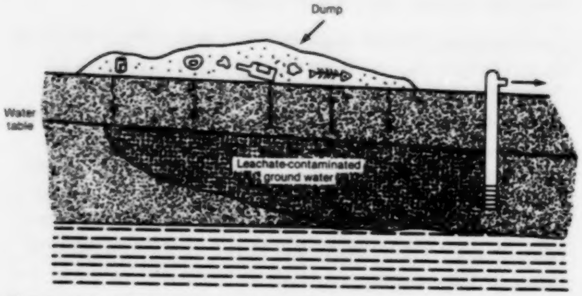


Figure 124. Ground-Water Contamination Caused By Leachate Infiltration from a Dump.

Disposal of Sewage and Water Treatment Plant Sludge. The sludge from treatment plants presents not only a significant waste disposal problem but one that is growing, significantly. The wastes include lime-rich sludge from water treatment plants as well as sewage sludge from waste water treatment plants. In recent years, municipal officials have attempted to solve the sewage-sludge problem by spreading the sludge on the land surface or filling abandoned strip mine pits with it. At first glance this may seem to be an effective means of disposal, but many exotic chemicals, derived from domestic, agricultural, municipal, and industrial wastes may exist in the sludge as soluble or relatively insoluble substances. When the sludges are used as fertilizers, the soluble compounds may infiltrate while the more insoluble compounds, many of which may consist of toxic metals, are removed and concentrated by plants. Much needs to be learned about the chemical and biological migration of numerous elements and compounds present in sludges.

Salt Spreading on Roads. In recent years, particularly since the construction of the interstate highway system, water pollution due to wintertime road salting has become an increasing problem. The effect is becoming even more severe as salt usage increases with a concomitant decrease in the use of sand. From a water quality viewpoint, the salting brings about deterioration of stream quality due to highly mineralized surface runoff, and the infiltration of briney water causes ground-water pollution.

Accidental Spills of Hazardous Materials. A large volume of toxic materials are transported throughout the country by truck, rail, and aircraft and are stored in aboveground tanks; accidential spills of these hazardous materials are not uncommon. There are virtually no methods that can be used to quickly and adquately clean up an accidental spill or spills caused by explosions or fires. Furthermore, immediately following an accident, the usual procedure is to spray the spill area with water. The resulting fluid may either flow into a stream or infiltrate the ground. In a few cases, the fluids have been impounded by dikes, which lead to even greater infiltration. In any case, water resources may be easily and irreparably polluted from accidental spills of hazardous materials.

Fertilizers and Pesticides. Increasing amounts of both fertilizers and pesticides are being used in the United States each year. Many of these substances are highly toxic and, in many cases, quite mobile in the subsurface. Many compounds, however, become quickly attached to fine-grained sediment, such as organic matter and clay and silt particles; a part of this attached material is removed by erosion and surface runoff. In many heavily fertilized areas, the infiltration of nitrate, a decomposition product of ammonia fertilizer, has grossly polluted ground water. The consumption of nitrate-rich water leads to a serious disease in infants commonly known as "blue babies" (methemoglobinemia).

In many irrigated regions, automatic fertilizer feeders attached to irrigation sprinkler systems are becoming increasingly popular. When the irrigation-well pump is shut off, water flows back through the pipe system into the well. This creates a partial vacuum in the lines that may cause fertilizer to flow from the feeder into the well. It is possible that some individuals are dumping fertilizers (and perhaps even pesticides) directly into the well to be picked up by the pump and distributed to the sprinkler system. In this case, it directly contributes to ground-water pollution.

Animal Feedlots. Animal feedlots cover relatively small areas but provide a huge volume of animal wastes. These wastes have polluted both surface and ground water with large concentrations of nitrate. Even small feedlots and liveries have created local but significant problems. In a few areas the liquid runoff from feedlots is collected in lined basins and pumped onto adjacent grounds as irrigation waters, providing a luxuriant growth.

Particulate Matter from Airborne Sources. A relatively minor source of ground-water pollution is caused by the fallout of particulate matter originating from smoke, flue dust, or aerosols. Some of the particulate matter is water-soluble and toxic. An example of this type of pollution is airborne chromium-rich dust that discharged through the roof ventilators of a factory in Michigan and accumulated on the downwind side of the plant. As Figure 125 shows, the highly soluble chromium compounds rapidly infiltrated and polluted a local municipal drinking water supply. Along the Ohio River at Ormet, Ohio, the airborne discharge of fluoride from an aluminum processing plant has seriously affected dairy operations and fluoride concentrations in ground water at the plant exceeded 1,000 mg/l in the mid 1970's.

Ground-Water Quality Problems that Originate in the Ground Above the Water Table

Many different types of materials are stored, extracted, or disposed of in the ground above the water table. Table 10 shows that water pollution can originate from many of these operations.

Septic Tanks, Cesspools, and Privies. Probably the major cause of ground water pollution in the United States is effluent from septic tanks, cesspools, and privies, although each site is small, as shown in Figure 126. Individually of little significance, these devices are important in the aggregate because they are so abundant and occur in every area not served by municipal or privately owned sewage treatment systems. The area that each point source affects is generally small, since the quantity of effluent is small, but in some limestone areas effluent may travel long distances in subterranean cavern systems.

Holding Ponds and Lagoons. The second major source of ground-water pollution is holding ponds and lagoons. As Figure 127 shows, these ponds and lagoons commonly consist of relatively shallow excavations that range in surface area from a few square feet to many acres. In some places they are euphemistically called "evaporation" ponds. Such ponds were commonly used to hold oil-field brines, and when the pond floors became sealed, the operators would disc them to increase infiltration. Holding ponds are also used to store municipal sewage as well as large quantities of wastes including a host of industrial chemicals. The latter are generally characterized by highly concentrated solutions that may contain toxic compounds.

Special problems develop with holding ponds and lagoons in limestone terrain where extensive near-surface solution openings have developed. In Florida, Alabama, Missouri, and elsewhere, municipal sewage lagoons have collapsed into sinkholes draining raw effluent into widespread underground openings. In some cases the sewage has reappeared in springs and streams several miles away. Wells producing from the caverns could easily become polluted and lead to epidemics of water-borne diseases.

Holding ponds are commonly considered to be liquid-tight but the vast majority leak. Although rarely reported, ground-water pollution caused by leaking holding ponds at a large number of industrial sites has been so extensive that all of the water supplies on the plant property are unusuable for many purposes without treatment. As a result, expensive treatment plants have been required. Moreover, the ground water may be so

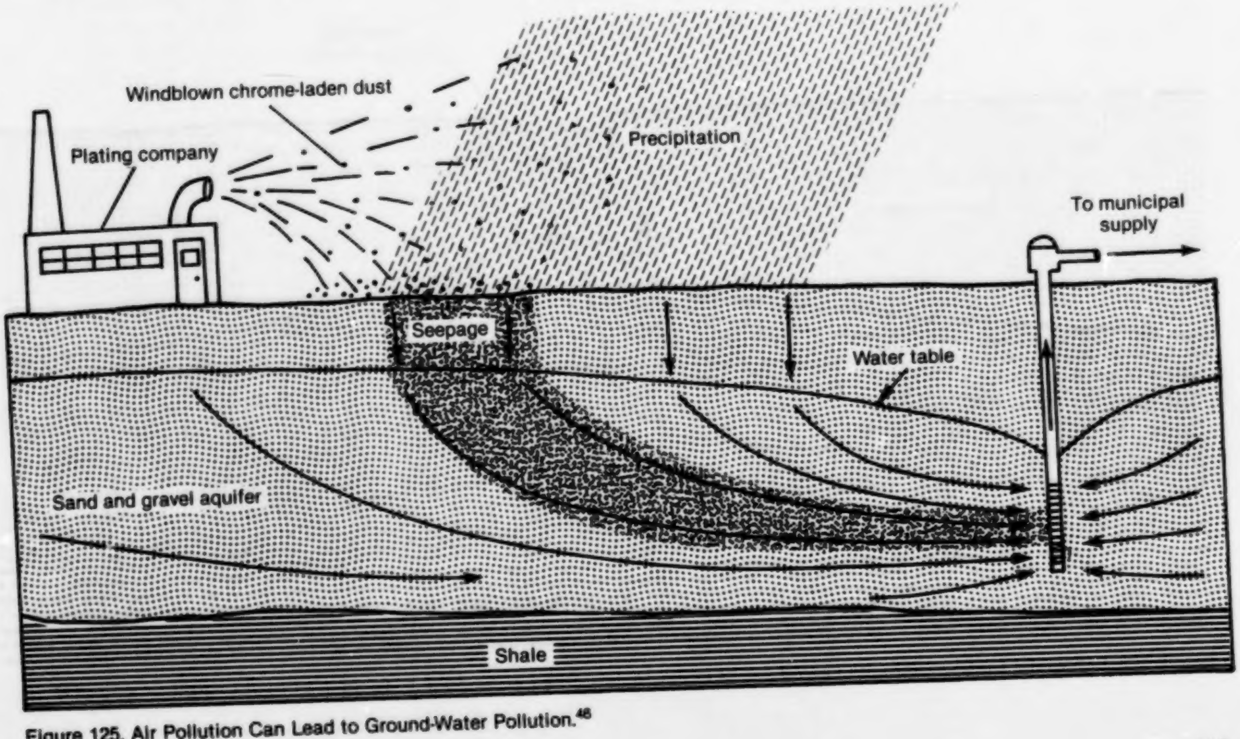


Figure 125. Air Pollution Can Lead to Ground-Water Pollution.[46]

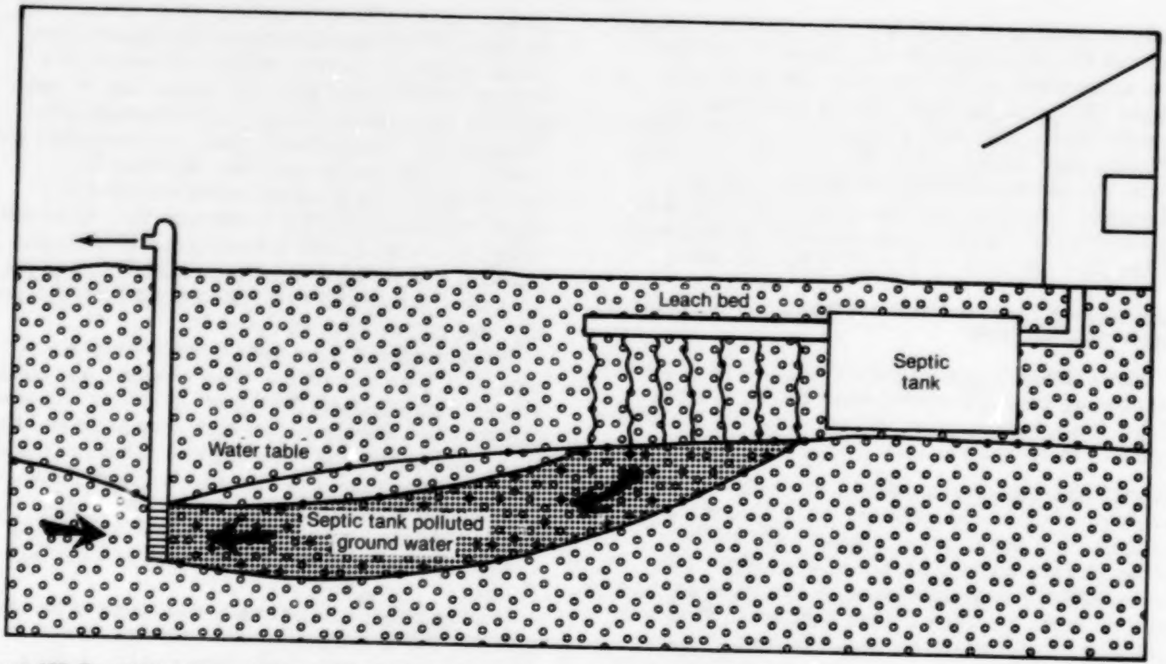


Figure 126. Percolation Through Zone of Aeration. Most of the Natural Removal or Degradation Processes Function Under These Conditions.

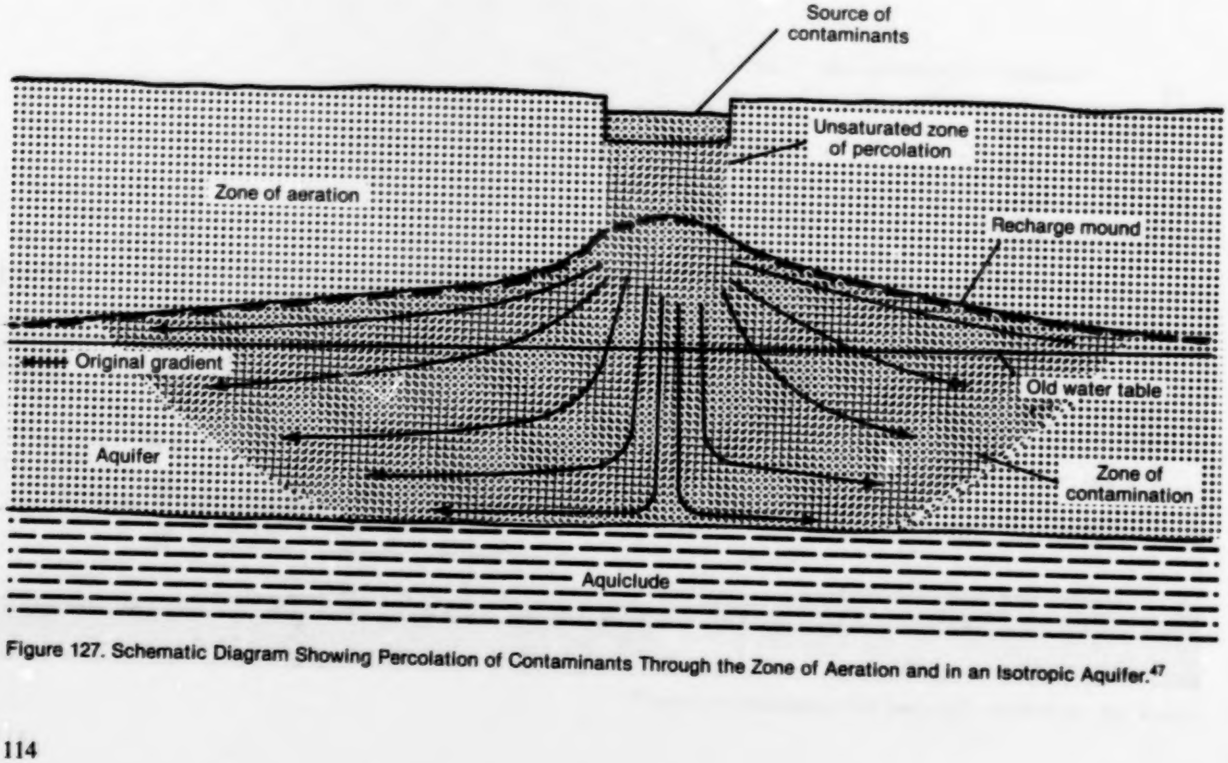


Figure 127. Schematic Diagram Showing Percolation of Contaminants Through the Zone of Aeration and in an Isotropic Aquifer.[47]

polluted that it exceeds surface water effluent standards and it cannot be pumped into adjacent streams.

Oil-field brines, a highly mineralized salt solution, are particularly noxious and without doubt they have locally polluted both surface and ground water in every state that produces oil. The brine, an unwanted byproduct, is produced with the oil. In many states it is disposed of by placing it in holding ponds from which it infiltrates into the ground. Commonly the oil well has been long abandoned before it becomes apparent that the adjacent ground water is polluted. This, in turn, may leave no possibility for recovery of damages by the landowner.

Sanitary Landfills. Sanitary landfills generally are constructed by placing wastes in excavations and covering the material daily with soil—thus the term "sanitary" to indicate that garbage and other materials are not left exposed to produce odors or smoke or attract vermin and insects. Even though a landfill is covered, however, leachate may be generated by the infiltration of precipitation and surface runoff. Fortunately many substances are removed from the leachate as it filters through the unsaturated zone, but leachate may pollute ground water and even streams if it discharges at the surface as springs and seeps.

At one site, rejected transformers and capacitors containing polychlorinated biphenyls from an industrial plant were disposed of in a municipal landfill. A number of stillbirths and birth defects soon occurred in cattle that drank water from a nearby stream. Analyses of the water showed large concentrations of PCB, the origin of which was, without question, the landfill.

Waste Disposal in Excavations. Following the removal of clay, limestone, sand, and gravel, or other material, the remaining excavations are traditionally left unattended and often are used as unregulated dumps. The quantity and variety of materials placed in dumps and excavations are almost limitless. Excavations also have been used for the disposal of liquid wastes, such as oil-field brines and spent acids from steel mill operations. Many other excavations serve as disposal sites for snow removed from surrounding streets and roads—snow that commonly contains a large amount of salt. Disposal of these and other wastes in excavations may lead to ground-water pollution.

Leakage from Underground Storage Tanks. A growing problem of substantial potential consequence is leakage from storage tanks and from pipelines leading to such tanks. Gasoline leakage has caused severe hazardous pollution problems throughout the nation. Gasoline floats on the ground-water surface and leaks into basements, sewers, wells, and springs, causing noxious odors, explosions, and fires. A single-wall steel tank has a life expectancy of 18 years and costs about $1 per gallon to replace. A cleanup operation will generally exceed $70,000.

Leakage from Underground Pipelines. Literally thousands of miles of buried pipelines crisscross the United States. Leaks, of course, do occur, but it may be exceedingly difficult to detect and locate them. Leaks are most likely to develop in lines carrying corrosive fluids. An example occurred in central Ohio where a buried pipeline carried oil-field brine from a producing well to a disposal well. The corrosive brine soon weakened the metal pipe, which then began to leak over a length of several tens of yards. The brine infiltrated, polluting the adjacent ground water, then flowed down the hydraulic gradient and discharged into a stream. During the ensuring months, nearly all of the vegetation between the leaking pipeline and the stream was killed. The leaking area of the pipe was detected only because of the dead vegetation and salty springs.

A vexing problem of chromium compounds that polluted several shallow wells in Michigan was traced to a leaky sewer transporting metal finishing wastes. Radioactive materials have also leaked from pipelines. The several leaks reported at the Hanford A.E.C. Works came about as a result of loaded, underground tanks settling differentially into the subjacent earth materials, causing the pipelines carrying radioactive waste to break at joints.

Artificial Recharge. Artificial recharge includes a variety of techniques used to increase the amount of water infiltrating an aquifer. It consists of spreading the water over the land or placing it in pits or ponds, from which the water will seep into the ground, or pumping water through wells directly into the aquifer. As water demands continue to increase, there is no doubt that artificial recharge will become more popular as a ground-water management tool.

Waters used for artificial recharge consist of storm runoff, excess irrigation water, stream flow, cooling water, and treated sewage effluent, among others. Obviously the quality of water artificially recharged can have a deleterious effect on the water in the ground under certain conditions.

Sumps and Dry Wells. Sumps and dry wells may locally cause some ground-water pollution and, in places where these structures are adjacent to a stream, bay, lake, or estuary, may pollute such surface-water bodies and perhaps lead to a proliferation of algae and other water weeds. These structures are commonly used to collect runoff or spilled liquids, which will infiltrate through the sump. Sumps and dry wells are typically installed to solve surface drainage problems, so they may transmit whatever pollutants are flushed into the well to ground water.

Graveyards. Leachate from graveyards may cause ground-water pollution, although cases are not well documented. In some of the lightly populated glaciated regions in the north central part of the United States, graveyards are commonly found on deposits of sand and gravel, because these materials are easier to excavate than the adjacent glacial till and, moreover, are better drained so that burials are not below the water table. Unfortunately, these same sand and gravel deposits may also serve as a major source of water supply. Graveyards are also possible sources of pollution in many hard rock terrains where there are sinkholes or a thin soil cover.

Ground-Water Quality Problems that Originate in the Ground Below the Water Table

Table 10 lists a number of major causes of ground-water pollution produced by the use and misuse of space in the ground below the water table.

Waste Disposal in Wet Excavations. Following the cessation of various mining activities, the excavations are commonly abandoned; eventually they may fill with water. These wet excavations have been used as dumps for both solid and liquid wastes. The wastes, being directly connected to an aquifer, may cause extensive pollution. Furthermore, highly concentrated leachates may be generated from the wastes due to seasonal fluctuations of the water table. In the late 1960's at a lead-zinc mine in northwestern Illinois, processing wastes were discharged into an abandoned mine working. The wastes, moving slowly in the ground water, polluted several farm wells. Analyses of water from several of the polluted wells showed high concentrations of dissolved solids, iron, sulfate, and, more importantly, heavy metals and cyanide.

Drainage Wells and Canals. Where surficial materials consist of heavy clay, flat-lying land may be poorly drained and contain an abundance of marshes and ponds. Drainage of this type of land is generally accomplished with field tiles and drainage wells. As Figure 128 shows, a drainage well is merely a vertical, cased hole in the ground or in the bottom of a pond that allows the water to drain into deeper, more permeable materials. The pond water may be polluted which, in turn, leads to deterioration of water quality in the receiving aquifer.

Deepening of stream channels may lower the water table. Where the fresh-saltwater interface lies at shallow depths, lowering of the water table (whether by channelization, pumping, or other causes) may induce upward migration of the saline water; it may even flow into the deepened channel and pollute the surface water, as Figure 129 shows. Under these circumstances, reduction of the depth to fresh water can result in a rise in the level of saline water several times greater than the distance the fresh water level is lowered.

In some coastal areas, particularly in Florida, the construction of extensive channel networks has permitted tidal waters to flow considerable distances inland. The salty tidal waters infiltrate, increasing the salt content of the ground water in the vicinity of the canal.

Well Disposal of Wastes. For decades, man has disposed of liquid wastes by pumping them into wells. Since World War II, a considerable number of deep well injection projects have come into existence, usually at industrial sites. Industrial disposal wells range in depth from a few tens of feet to several thousand feet. The injection of highly toxic wastes into come of these wells has led to several water-pollution problems. The problems are caused by the pollution of fresh water due to direct injection into the aquifer as well as leakage of pollutants from the well head, through the casing, or via fractures in confining beds. Injection of liquid wastes near Denver by means of deep well disposal apparently caused an increase in the frequency of local earthquakes. Deep well injection in the vicinty of Sarnia, Ontario, caused several long-abandoned brine wells in Michigan to flow because of the greatly increased aquifer pressure.

Exclusive of oil-field brine, most deep well injection operations are tied to the chemical industry. Well depths range from 1,000 to 9,000 feet and average 4,000 feet. The deepest wells are found in Texas and Mississippi.

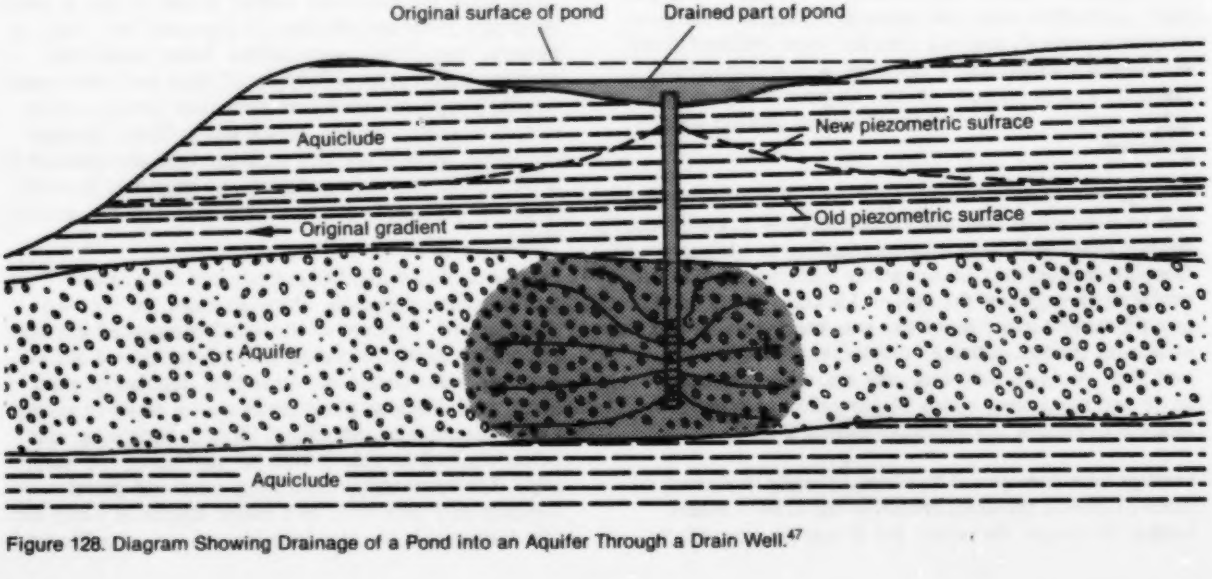


Figure 128. Diagram Showing Drainage of a Pond into an Aquifer Through a Drain Well.[47]

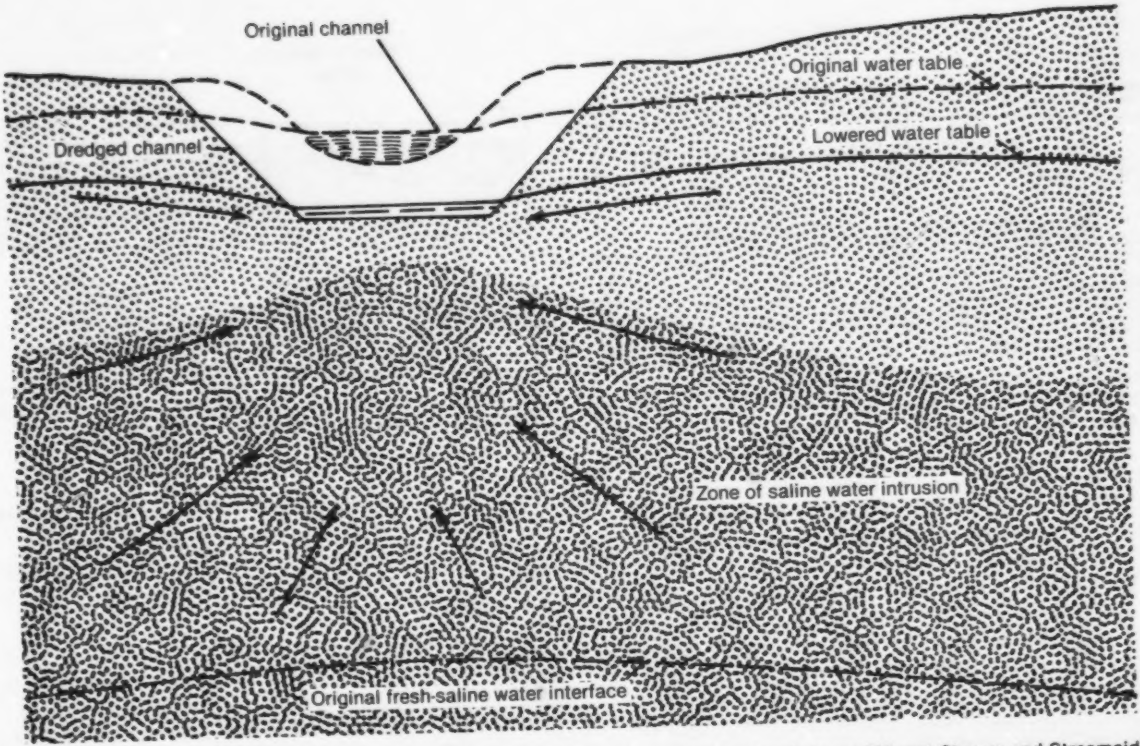


Figure 129. Diagram Showing Migration of Saline Water Caused by Lowering of Water Levels in an Effluent Stream and Streamside Aquifer Hydraulically Connected to an Underlying Saline Water Aquifer.[48]

As of October 1983, EPA reported the existence of at least 188 active hazardous waste injection wells in the United States. There was an additional 24,000 wells used to inject oil-field brine.

Properly managed and designed underground injection systems can be effectively used for storage of wastes deep underground and may permit recovery of the wastes in the future. Before deep well disposal of wastes is permitted by EPA, however, there must be an extensive evaluation of the well system design and installation, the waste fluids, and the rocks in the vicinity of the disposal well.

Underground Storage. The storage of material underground is attractive from both economic and technical viewpoints. Natural gas is one of the most common substances stored in underground reservoirs. However, the hydrology and geology of underground storage areas must be well understood in order to insure that the materials do not leak from the reservoir and degrade adjacent water supplies.

Secondary Recovery. With increased demands for energy resources, secondary recovery, particularly of petroleum products, is becoming even more important. Methods of secondary recovery of petroleum products commonly consist of injection of steam or water into the producing zone, which either lowers the viscosity of the hydrocarbon or flushes it from the rocks, enabling increased production. Unless the injection well is carefully monitored and constructed, fluids can migrate from a leaky casing or through fractures in confining units.

Mines. Mining has caused a variety of water pollution problems. These problems are caused by pumping of mine waters to the surface, by leaching of the spoil material, by waters naturally discharging through the mine, and by milling wastes, among others. Literally thousands of miles of stream and hundreds of acres of aquifers have been polluted by highly corrosive mineral-

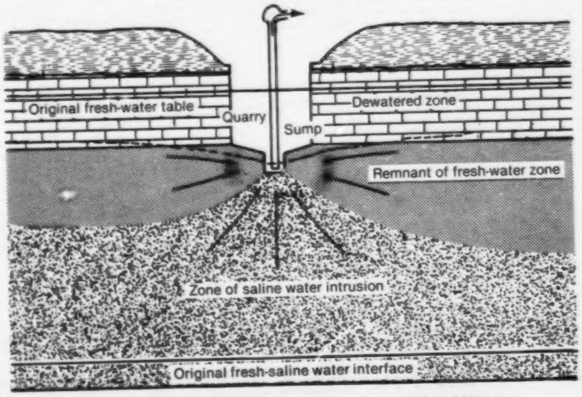

Original fresh-water table
Quarry
Sump
Dewatered zone
Remnant of fresh-water zone
Zone of saline water intrusion
Original fresh-saline water interface

Figure 130. Diagram Showing Migration of Saline Water Caused by Dewatering in a Fresh-Water Aquifer Overlying a Saline-Water Aquifer.[48]

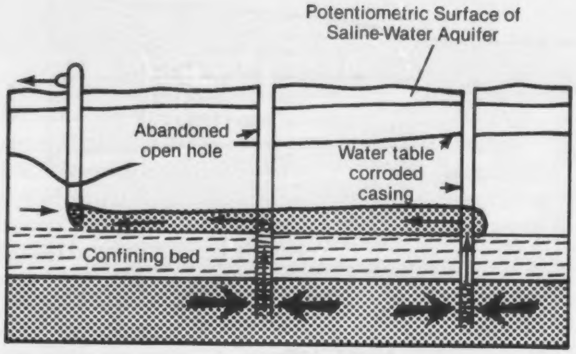

Potentiometric Surface of Saline-Water Aquifer
Abandoned open hole
Water table corroded casing
Confining bed

Figure 131. Upward Leakage and Flow Through Open Holes. Some Important Aquifers Have Been Ruined by Improper Drilling Practices.

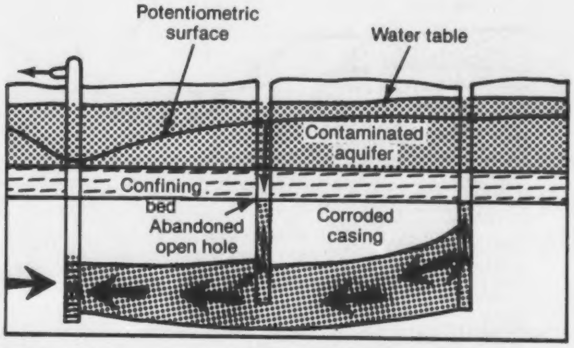


Figure 132. Downward Leakage. Contamination of One Aquifer Can Affect Others in a Multi-Aquifer System.

ized waters originating in coal mines and dumps in Appalachia. In many western states, mill wastes and leachates from metal sulfide operations have seriously affected both surface water and ground water.

Many mines are deeper than the water table, and in order to keep them dry, large quantities of water are pumped to waste. If salt water lies at relatively shallow depths, Figure 130 shows that the pumping of freshwater for dewatering purposes may cause an upward migration of the salt water, which may be intercepted by the well. The mineralized water most commonly is discharged into a surface stream.

Exploratory Wells and Test Holes. Literally hundreds of thousands of abandoned exploratory wells dot the countryside. Many of these holes were drilled to determine the presence of underground mineral resources (seismic shot holes, coal, salt, oil, gas, etc.). The open holes permit water to migrate freely from one aquifer to another. As Figures 131 and 132 show, a freshwater aquifer could thus be joined with a polluted aquifer or a deeper saline aquifer, or polluted surface water could drain into freshwater zones.

Abandoned Wells. Another major cause of ground-water pollution is the migration of mineralized fluids through abandoned wells. In many cases when a well is abandoned the casing is pulled (if there is one) or the casing may become so corroded that holes develop. This permits ready access for fluids under higher pressure to migrate either upward or downward through the abandoned well and pollute adjacent aquifers. In other cases, improperly cased wells allow high-pressure artesian saline water to spread from an uncased or partly cased hole into shallower, lower-pressure aquifers or aquifer zones, resulting in widespread salt intrusion.

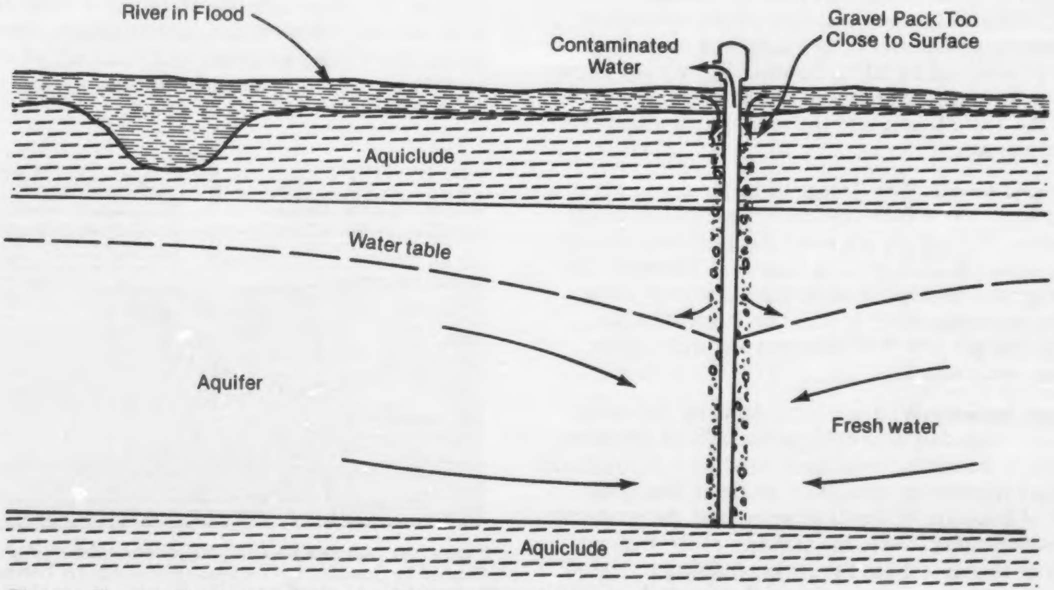
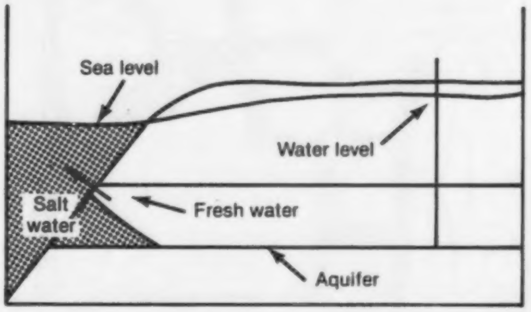


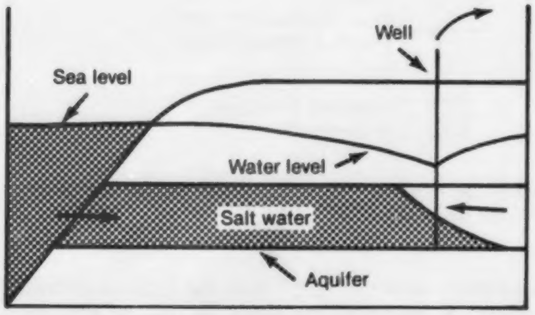
Figure 133. Diagram Showing Flood Water Entering a Well Through an Improperly Sealed Gravel Pack.

Water-Supply Wells. Improperly constructed water-supply wells may either pollute an aquifer or produce polluted water. Dug wells, generally of large diameter and shallow depth, and poorly protected, are commonly polluted by surface runoff flowing into the well. As Figure 133 illustrates, other pollution has been caused by infiltration of water through polluted fill around a well or through the gravel pack. Still other pollution has been caused by barnyard, feedlot, septic tank, or cesspool effluent draining directly into the well. Many pollution and health problems can arise because of poor well construction.

Ground-Water Development. In certain situations pumping of ground water can induce significant water-quality problems. The principal causes include interaquifer leakage, induced infiltration, and landward migration of sea water in coastal areas. In these situations the lowering of the hydrostatic head in a freshwater zone leads to migration of more highly mineralized water toward the well site. Undeveloped coastal aquifers are commonly full, the hydraulic gradient slopes towards the sea, and freshwater discharges from them through springs and seeps into the ocean, as shown in Figure 134. Extensive pumping lowers the fresh-water potentiometric surface permitting sea water to migrate toward the pumping center. Figure 135 shows a similar predicament which occurs in inland areas where saline water is induced to flow upward, downward, or laterally into a fresh water aquifer due to the decreased head in the vicinity of a pumping well. Wells drilled adjacent to streams induce water to flow from the streams to the wells. If the stream is polluted, induced infiltration will lead to deterioration of the water quality in the aquifer.



A. Natural Conditions



B. Sea-water Intrusion

Figure 134. Sea-Water Intrusion is Caused by Overpumping of Coastal Aquifers.

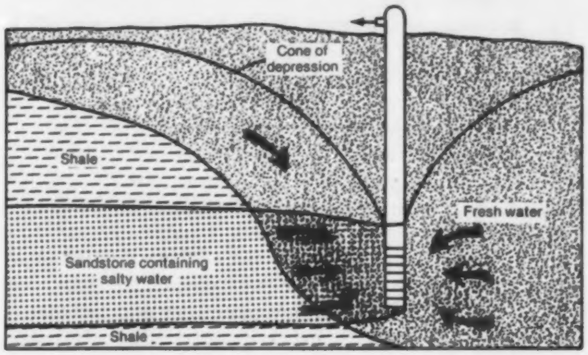


Figure 135. Diagram Showing How a Pumping Well Can Induce Highly Mineralized Water to Flow from a Saline Aquifer into a Fresh-Water Aquifer.

Natural Controls on Ground-Water Contamination

As Deutsch[47] clearly pointed out, there are four major natural controls involved in shallow ground-water contamination. The first control includes the physical and chemical characteristics of the earth materials through which the liquid wastes flow. A major attenuating effect for many compounds is the unsaturated zone, which has been called the "living filter". Many chemical and biological reactions in the unsaturated zone lead to contaminant degradation, precipitation, sorption, and oxidation. The greater the thickness of the unsaturated zone, the more attenuation is likely to take place. Below the water table, the mineral content of the medium probably becomes more important because various clays, hydroxides, and organic matter take up some of the contaminants by exchange or sorption. Many of the other minerals may have no effect on the contaminants with which they come into contact.

The second major control includes the natural processes that tend to remove or degrade a contaminant as it flows through the subsurface from areas or points of recharge to zones or points of discharge. These processes include filtration, sorption, ion-exchange, dispersion, oxidation, and microbial degradation, as well as dilution.

The third natural control relates to the hydraulics of the flow system through which the waste migrates, beginning with infiltration and ending with discharge. The contaminant may enter an aquifer directly, by flowing through the unsaturated zone, by interaquifer leakage, by migration in the zone of saturation, or by flow through open holes.

The final control is the nature of the contaminant. This includes its physical, chemical, and biological characteristics and, particularly, its stability under varying conditions. The stability of the more common constituents and the heavy metals are fairly well known although more complex than commonly realized. On the other hand, the stability of organic compounds, particularly synthetic organic compounds, has only recently come under close inspection and actually little is known of their degradation and mobility in the subsurface. This fact has been brought clearly to the attention of the general public by the abundance of reported incidences of ground-water contamination by EDB, TCE, and DBCP.

To a large extent, it is the aquifer framework that controls the movement of ground water and contaminants. Of prime importance, of course, is the hydraulic conductivity, both primary and secondary. In the case of consolidated sedimentary rocks, primary permeability, or that which came into being with the formation of the rock, in many respects is more predictable than secondary permeability, which came later and includes fractures and solution openings, among others. In sedimentary rocks similar units of permeability tend to follow bedding planes or formational boundaries, even if the strata are inclined. Permeable zones are most often separated by layers of fine-grained material, such as clay, shale, or silt, which serve as confining beds. Although leakage through confining beds is the rule rather than the exception, both water and contaminants are more likely to remain in a permeable zone than to migrate through a thickness of several units of different permeability. The movement of ground water and contaminants through larger openings, such as fractures, complicates the assumed picture. Figure 136 illustrates this movement. Not only can the velocity change dramatically, but in fracture flow, much of the attenuation capacity is lost, and it is difficult to predict local directions of flow.

The geologic framework, in conjunction with surface topography, also exerts a major control on the configuration of the water table and the thickness of the unsaturated zone. Generally speaking, a deposit of permeable surficial sand and gravel would be characterized by a water table that is relatively flat. In contrast, a covering of glacial till, which is typically fine-grained, would be characterized by a water-table surface that more closely conforms to the elevation of the land surface. The position of the water table is important not only because it is the boundary between the saturated and unsaturated zones, but also because it marks the bottom and, therefore, the thickness of the unsaturated material.

In many, if not most, contaminated areas, the water table has been or is intermittently affected by pumping. The resulting cone of depression on the water table changes both the hydraulic gradient and ground-water velocity resulting in flow to a discharging well. A change in gradient and velocity also occurs in the vicinity of recharge basins (lagoons, pits, shafts. etc.), because the infiltrating water forms a mound in the water table. As Figure 137 shows, the mound causes radial flow and, therefore, contaminants will move in directions that are different than the regional hydraulic gradient, at least until the mounding effects are overcome and the regional gradient is reestablished.

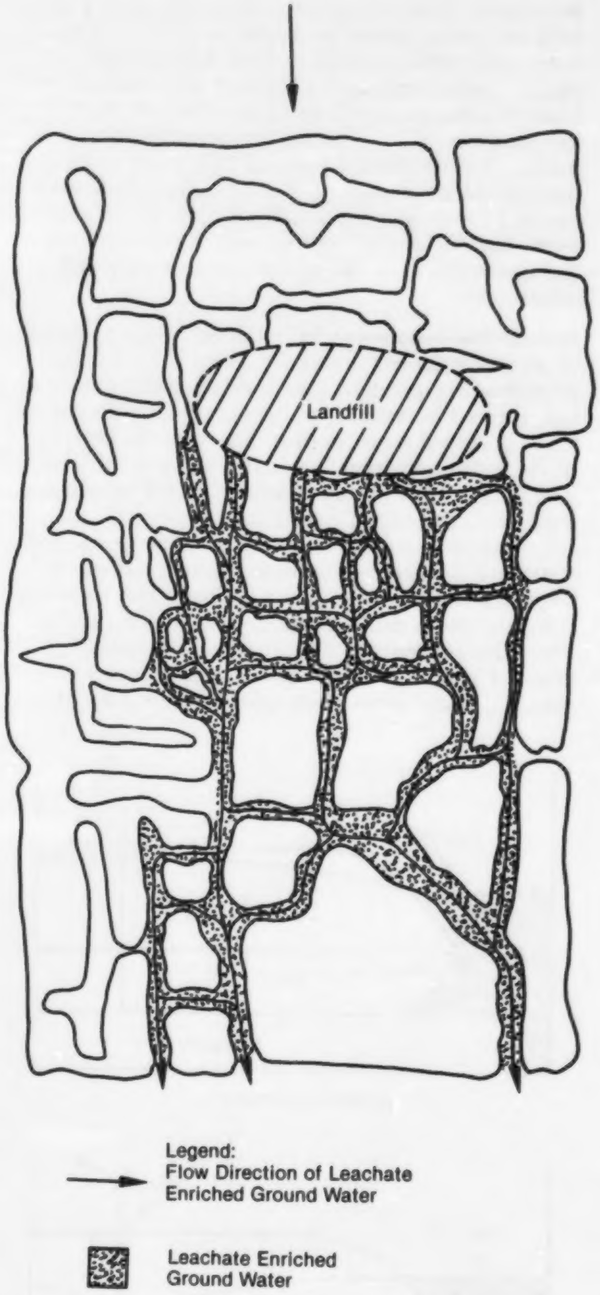

Landfill
Legend:
Flow Direction of Leachate Enriched Ground Water
Leachate Enriched Ground Water

Figure 136. Movement of Ground-Water and Contaminants Through Large Openings.

Ground-water or interstitial velocity is controlled by the hydraulic conductivity, gradient, and effective porosity. Water movement through a permeable gravel

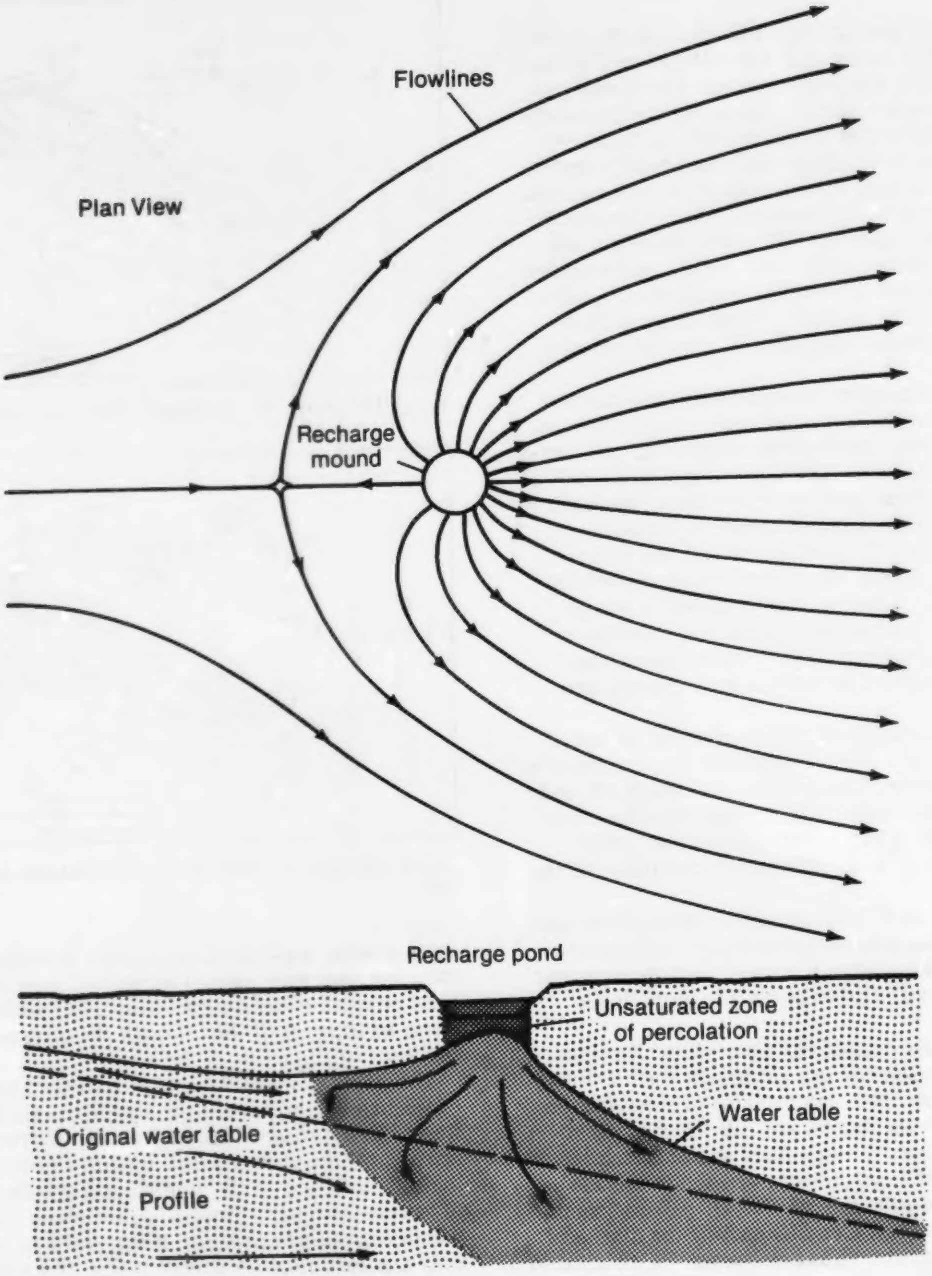


Figure 137. Diagrams Showing Lines of Flow from a Recharge Mound on a Sloping Water Table.

with a gradient of 10 feet per mile averages about 60 feet per day, but in a clay with the same gradient and no secondary permeability the water movement would be only about 1 foot in 30,000 years. In most aquifers, ground-water velocity ranges from a few feet per day to a few feet per year.

Carlston and others[49] determined that the mean residence time of ground water in a basin in Wisconsin was about 45 days and in New Jersey about 30 days. This study shows that ground water may discharge into closely spaced streams in humid areas within a few days to a few months. On the other hand, in less permeable terrains ground water and contaminants may remain in the ground for years or even decades.

Leachate

The causes of ground-water pollution are many, but it is the source that needs special consideration. For example, an accidental spill from a ruptured tank may provide a considerable volume of liquid with an extremely high concentration that is present only during a very short time span, but leachate generated from a landfill may consist of a large volume with a low concentration that spans a period of many years. Once it reaches the water table, the accidental spill might move as a conservative contaminant because of its high concentration despite the fact that it might be degradable in smaller concentrations. The leachate is more likely to be attenuated by microbial degradation, sorption, dilution, and dispersion.

In the case of landfills, leachate is a liquid that has formed as infiltrating water migrates through the waste material extracting water-soluble compounds and particulate matter. The mass of leachate is directly related to precipitation, assuming the waste lies above the water table. Much of the annual precipitation, including snowmelt, is removed by surface runoff and evapotranspiration; it is only the remainder that is available to form leachate. Since the landfill cover, to a large extent, controls leachate generation, it is exceedingly important that the cover be properly designed, maintained, and monitored in order to minimize leachate production.

Schuller and others[50] described the effect of regrading, installation of a PVC topseal and revegetation of a landfill in Windham, Connecticut. As Figures 138 and 139 illustrate, field data clearly indicate that the cover reduced infiltration and leachate generation, which caused a reduction in the size and concentration of the leachate plume.

According to an EPA estimate[51], a disposal site consisting of 17 acres with 10 inches/year of infiltration could produce 4.6 million gallons of leachate each year for 50 to 100 years. This estimate, of course, is site dependent.

Feen and others[52] described a landfill in Cincinnati, Ohio, and estimated leachate generation using a water balance technique. The data in Table 11 show that percolation through the landfill was calculated to occur only during January through April and in December. In this 50 acre landfill, the leachate generation averaged about 11,394,000 gallons/year, 949,475 gallons/month, or 31,216 gallons/day. Considering only the calculated values for the months when precolation was assumed to occur, leachate generation averaged about 113,900 gallons/day in January, 120,740 in February, 96,400 in March, 31,700 in April, and 17,500 gallons/day in December, in contrast to the annual daily average of 31,216 gallons.

The Cincinnati calculations appear unusual because the spring runoff in Ohio normally occurs during March and April and, since there is a soil-moisture excess at that time, most ground-water recharge takes place during this interval. Moreover, ground-water recharge may occur any time there is rain. Therefore, one must use caution when applying water balance techniques to estimate leachate generation. The method may provide a good estimate or long-term average, but it likely produces an estimate of the total volume of recharge that is too low.

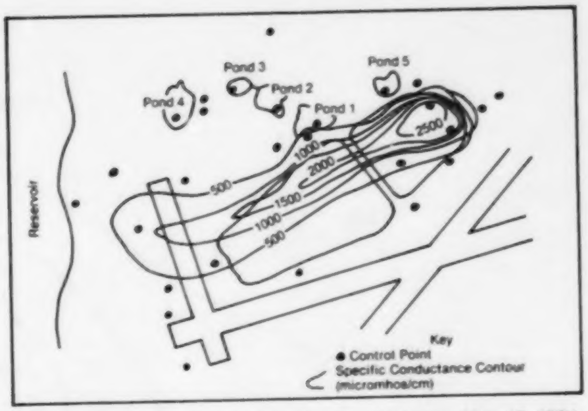


Figure 138. Distribution of Specific Conductance, May 19, 1981.

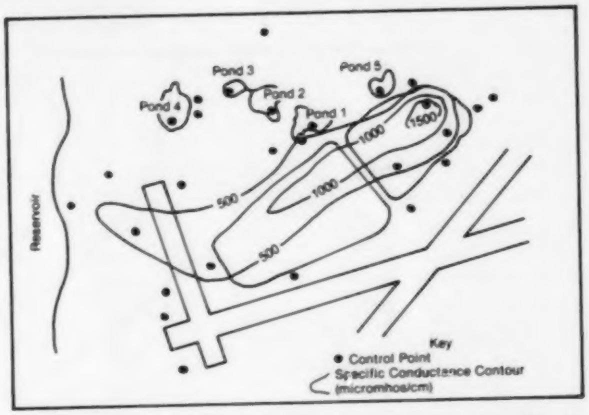


Figure 139. Distribution of Specific Conductance, November 12, 1981.

The physical, chemical, and biological characteristics of leachate are influenced by: (1) the composition of the waste, (2) the stage of decomposition, (3) the microbial activity, (4) the chemical and physical characteristics of the soil cover and of the landfill, and (5) the time rate of release (recharge). Since all of the above can range within extremely wide limits, it is possible to provide only a general range in concentration of leachate constituents, as Table 12 shows.

It is also important to account for the fact that materials placed in landfills may range widely depending on the season. For example, many municipal landfills are used to dispose of snow and ice, which may contain large concentrations of calcium, sodium, and chloride from de-icing salts. This could lead to the generation of leachate that varies seasonally, particularly in regard to the chloride concentration. It is also important to remember that leachate collected from a seep at the base of a landfill should be more highly mineralized than the

Parameter*	J	F	M	A	M	J	J	A	S	O	N	D	Annual
PET	0	2	17	50	102	134	155	138	97	51	17	3	766
P	80	76	89	82	100	106	97	90	73	65	83	84	1025
$C_{R/O}$	0.17	0.17	0.17	0.17	0.17	0.13	0.13	0.13	0.13	0.13	0.13	0.17	
R/O	14	13	15	14	17	14	13	12	9	8	11	14	154
I	66	63	75	68	83	92	84	78	64	57	72	70	872
I-PET	+66	+61	+58	+18	-19	-42	-71	-60	-33	+6	+55	+67	+106
ΣNEG (I-PET)				(0)	-19	-61	-132	-192	-225				
ST (Table C)	150	150	150	150	131	99	61	41	33	39	94	150	
ΔST	0	0	0	0	-19	-32	-38	-20	-8	+6	+55	+56	
AET	0	2	17	50	102	124	122	98	72	51	17	3	658
PERC	+66	+61	+57	+18	0	0	0	0	0	0	0	+11	213

*The parameters are as follows: PET, potential evapotranspiration; P, precipitation; $C_{R/O}$, surface runoff coefficient; R/O, surface runoff, I, infiltration; ST, soil moisture storage; ΔST, change in storage; AET, actual evapotranspiration; PERC, percolation. All values are in millimeters (1 inche = 25.4 mm).

Table 11. Water Balance Data for Cincinnati, Ohio.

Constituents	Operating Landfill	Abandoned Landfill
BOD_5, mg/l	1,800	18
COD, mg/l	3,850	246
Ammonia-N, mg/l	160	100
Hardness, mg/l as $CaCO_3$	900	290
Total Iron, mg/l	40.4	2.2
Sulfate, mg/l	225	100
Specific Conductance, mmhos	3,000	2,500

Table 12. Comparison of Chemical Characteristics of Leachate from an Operating Landfill and a 20-Year-Old Abandoned Landfill in Southeastern Pennsylvania.[53]

leachate present in the underlying ground water, which is diluted.

Cyclic Changes in Ground-Water Quality

It is commonly assumed and often reported that natural ground-water quality is nearly constant at any particular site. Field data substantiate this assumption, and logic leads to the same conclusion if the aquifer is not submitted to a new stress. On the other hand, multiple samples from a single well are very likely to show slight changes in concentrations of specific constituents, due to differences in sample collection, storage, and analytical technique.

Conversely, investigators are finding with increasing frequency that ground-water quality, at least in shallow or surficial aquifers, can change significantly and rather rapidly, perhaps as much as an order of magnitude within a few hours or days, even though there is no source of man-made contamination.

Deeper or confined aquifers generally are characterized by a nearly constant chemical quality that, at any particular site, reflects the geochemical reactions that occurred as the water migrated through confining layers and aquifers from its recharge area to the point of collection or discharge.

The quality of deeper water can change, but generally not abruptly, in response to stresses on the aquifer system. Changes in hydrostatic head brought about by pumping, for example, may cause leakage of more highly mineralized water from adjacent units into the producing zone. This leakage may be due to fluid migration along the well casing or gravel pack or by leakage through confining beds or abandoned wells or exploration holes. Another cause of chemical change is waste disposal, particularly through well injection.

Surficial or shallow aquifers, however, are not well protected from chemical changes brought about by natural events occurring at the land surface or from man-induced pollution. Surficial aquifers are, in fact, the most susceptible to rapid and sometimes dramatic changes in quality. Some of the changes are related to man's activities; others are not.

The Concept of Cyclic Fluctuations

Several years ago Pettyjohn[29, 54] described cyclic fluctuations of ground-water quality. The mechanisms that

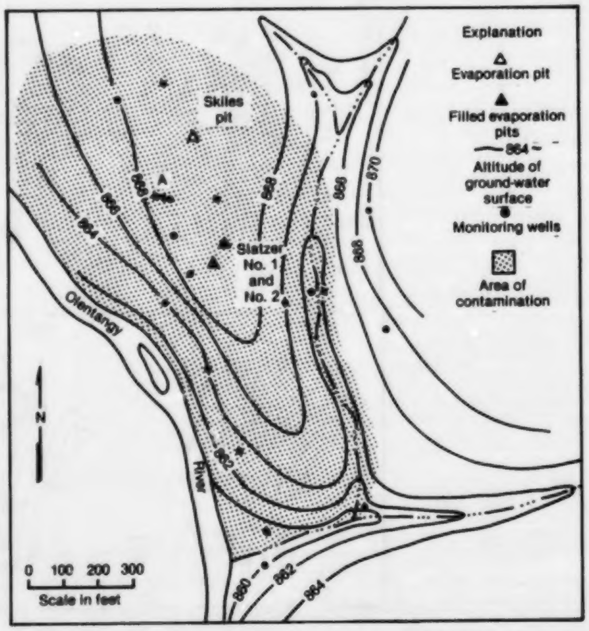


Figure 140. Water-Level Surface and Monitoring Wells at the Contaminated Site.

lead to cyclic fluctuations will be discussed in greater detail here because both the cause and effect can have a significant impact on: (1) ground-water quality monitoring and determination of background quality, (2) transport and fate of organic and inorganic compounds, as well as bacteria and viruses, and (3) monitoring well design and installation.

The site that Pettyjohn[54] used to develop the concept of cyclic fluctuation was a particularly well-instrumented area in central Ohio. The contamination area, shown in Figure 140, lies on the floodplain of the Olentangy River and is bounded on the east and south by small intermittent streams and on the west by the river. Underlain by shale, the alluvial deposits consist of sand, gravel, silt, and clay, that range from 15 to 35 feet in thickness. The water table lies from 1.5 to 5 feet below land surface and fluctuates a maximum of a foot or so throughout the year. Precipitation averages about 38 inches per year.

Oil production began at this site in mid-1964, but by July 1965, all wells had been plugged. Ground-water contamination occurred because of leakage of oil-field brine from three holding ponds. One pond (Skiles) was used from July 1, 1964 to June 30, 1965; about 126 barrels of brine were placed in it. Two other ponds (Slatzer), received 110,000 barrels of brine containing about 35,000 mg/l of chloride over approximately the same time period.

When samples were first collected from 23 monitoring wells in July 1965, the aquifer locally contained more than 35,000 mg/l of chloride. At this time the ponds were abandoned and the two Slatzer pits were filled with previously excavated materials that had formed surrounding embankments. The fact that the ground water contained higher concentrations of chloride than the original brine is most likely due to increased concentrations in the holding ponds brought about by evaporation.

As Figure 141 shows, an interesting relationship becomes apparent when examining the areal extent of ground-water contamination with time. Note that the area enclosed by the 1,000 mg/l isochlor during 1965–66 changed monthly but the changes did not necessarily encompass smaller areas. The 1969 data show a similar phenomenon. This suggests that a linear flushing rate did not exist.

Of particular importance in the monitoring of this site are three adjacent wells, one screened at a depth of 9 feet and another at 23 feet, while a third is gravel-packed through much of its length (23 feet) and receives water from the entire aquifer. Figure 142 illustrates these three wells. The locations of these wells are shown in Figure 140 at the position marked A. It is assumed that the first two wells represent the quality that exists at depths of about 7 to 9 feet and 21 to 23 feet, respectively, and that the gravel-packed well provides a composite sample of the reservoir. It is also assumed that when the composite well had a higher concentration than both the deep and shallow wells, the most highly mineralized water was between 9 and 23 feet.

Figure 143 shows the chloride fluctuations in the three wells during 1965, 1966, and 1969. Notice that at certain times the highest concentrations occur at the shallowest depths, at other times at the greatest depth, and at still

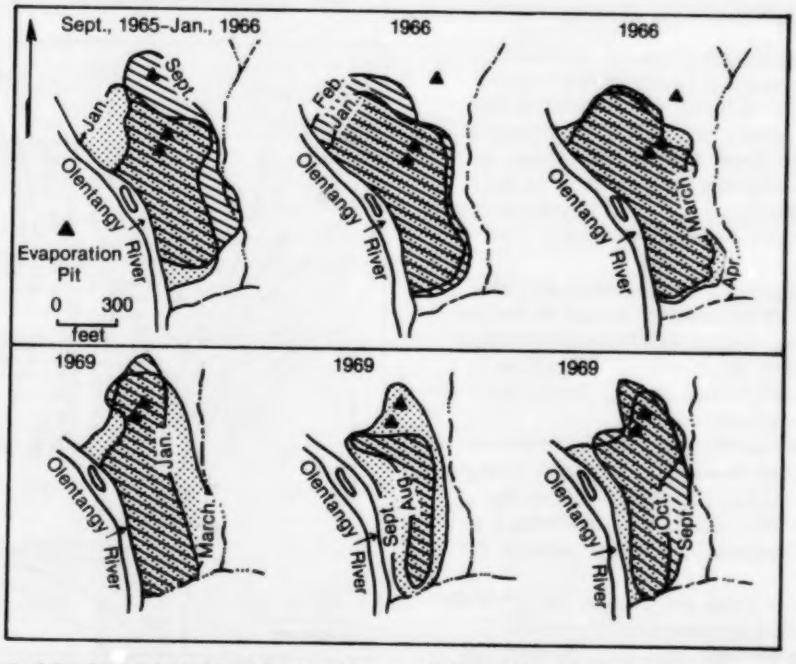


Figure 141. Areal Extent of the Contaminated Area Enclosed by the 1,000 mg/l (1965, 1966) and the 500 mg/l Isochlors During Selected Months. Contours Based on Data from Monitoring Wells.

other times the greatest concentration must lie somewhere in the middle of the aquifer. The only means for accounting for the variable distribution is intermittent reintroduction of the contaminant, which is puzzling in view of the fact that oil-field activities ceased in June 1965 before any of these samples were collected.

Figures 144 and 145 show another technique for illustrating the temporal-vertical distribution of chloride in the aquifer. These illustrations are based on monthly data obtained from the three adjacent wells. Concentrations at depths of 9 and 23 feet were measured; interpretation of the chloride distribution between these points was based on data from the fully penetrating well. In October and November 1965, the highest chloride concentrations were present at the shallowest depth but from December 1965 to April 1966 the highest concentrations were near the bottom of the aquifer. Furthermore, in November and December, the water in the middle of the aquifer was less mineralized than that above or below.

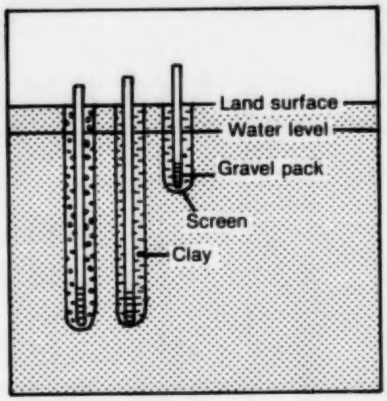


Figure 142. Completion Details of Three Closely Spaced Monitoring Wells.

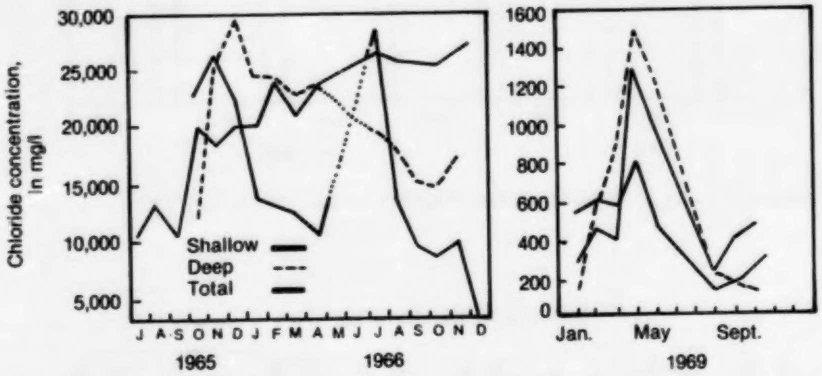


Figure 143. Fluctuation of Chloride Content in Closely Spaced Wells of Different Depths During 1965, 1966, and 1969.

Although greatly reduced, in January 1969 the largest concentration of chloride was again at the shallowest depth, but the situation was reversed during April and May. During February and March the central part of the aquifer was less mineralized than adjacent parts. By August there was only a slight chloride increase with depth, but in September and October the greatest concentrations again appeared in the central part of the aquifer.

The chloride fluctuations that occurred during 1965–66 and 1969 are shown schematically in Figure 146. The October 1965 samples apparently were collected shortly after a recharge event, which leached salt from the ground and formed a highly concentrated mass. This slowly sinking mass (1) was subsequently replaced with less mineralized water. A month later, the first mass had reached and was migrating along the bottom of the aquifer when another recharge event occurred (2). By December, the second mass had reached the bottom of the aquifer and was moving toward the river. Recharge events also occurred in January 1966 (3) and in February 1966 (4). The February event represented the spring runoff when evapotranspiration was minimal and the soil-moisture content exceeded field capacity over a wide area. This major period of recharge caused a large influx of salt and by March, the aquifer was contaminated throughout a wide area. This mass eventually discharged into the river.

In spite of the fact that brine disposal ceased by mid-1965, Figure 146 shows that the aquifer was recontaminated several times during 1969. Following an established pattern, small recharge events took place in January, February, and March 1969.

This study indicates that water soluble substances on the land surface or in the unsaturated zone may be intermittently introduced into a shallow aquifer for many years. The introduction of these contaminants is dependent upon the chemistry of the waste and the soil and upon the frequency of the recharge events. These recharge events are controlled by evapotranspiration, by the rate, duration, and intensity of precipitation, and by soil-moisture conditions.

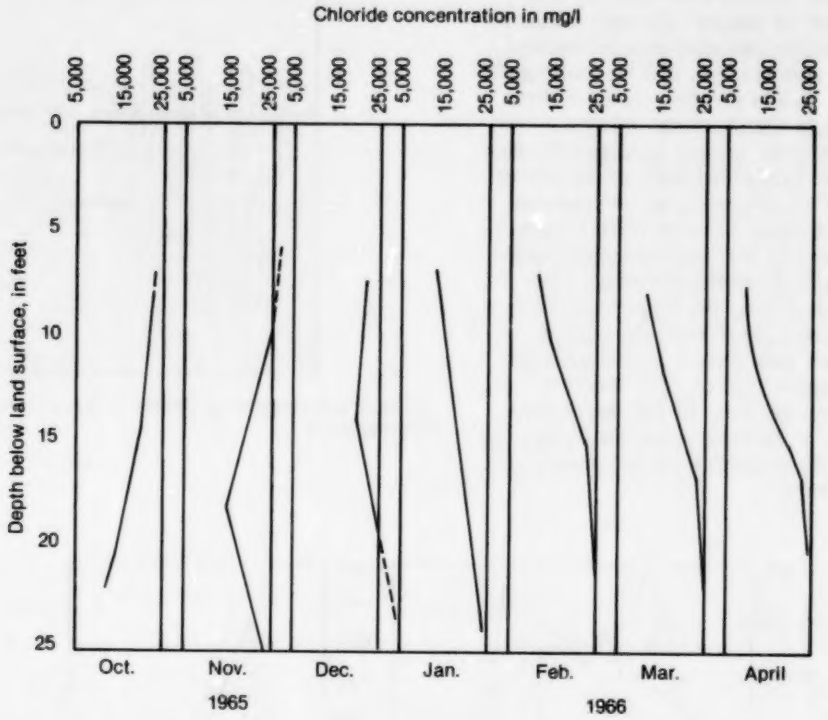


Figure 144. Vertical Distribution of Chloride at the Contaminated Site from October 1965 to April 1966

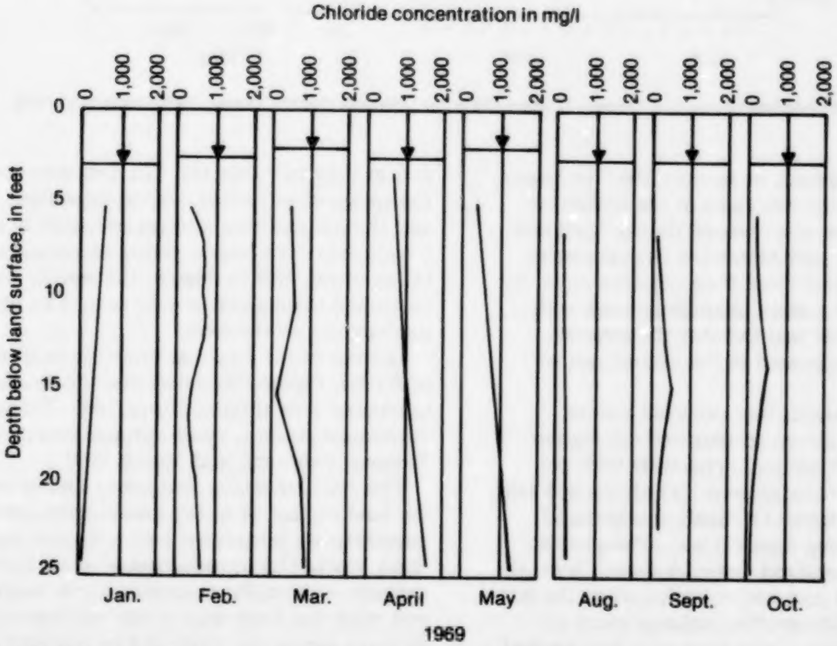


Figure 145. Vertical Distribution of Chloride at the Contaminated Site During 1969

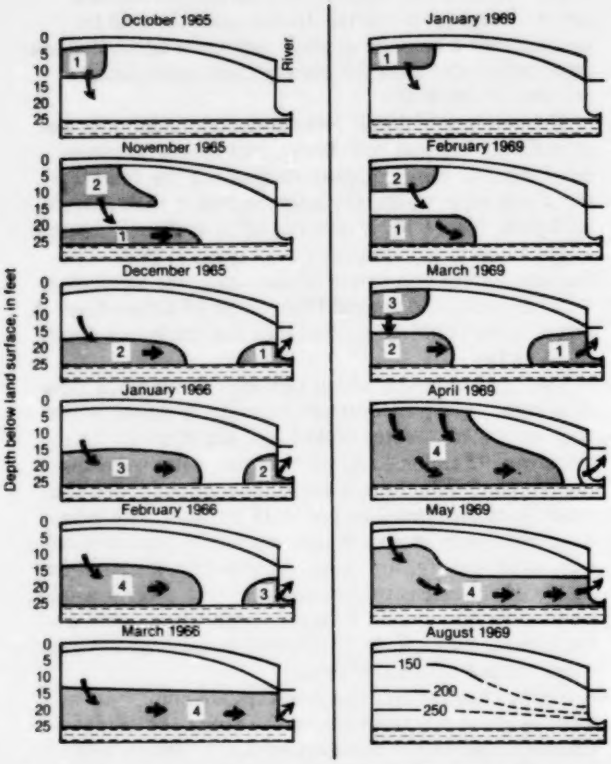


Figure 146. Schematic Diagram Showing the Cyclic Movement of Masses of Contaminated Water Through the Aquifer During Selected Months in 1965, 1966, and 1969. Stippled Areas for 1965-66 Represent Concentrations in Excess of 20,000 mg/l and for 1969 Represent Concentrations Greater than 500 mg/l.

Throughout most of the year in humid and semiarid regions, the quantity of water that infiltrates and the amount of contaminants that are washed into an aquifer are relatively small. On the other hand, during the spring recharge period significant quantities of contaminants may be flushed into the ground over wide areas. Therefore, the major influx of contaminants occurs on an annual basis, although minor recharge events may occur at any time.

In order to account for cyclic fluctuations in groundwater quality it is assumed that: (1) the unsaturated zone may store a considerable volume of water-soluble substances for long periods of time, and (2) the main paths along which contaminants move through the unsaturated zone to the water table consist largely of fractures and macropores.

Large concentrations of many water-soluble substances are stored within the unsaturated zone. Nitrate storage has been extensively studied. For example, in parts of western Kansas, the top 3 to 4 feet of unplowed prairie soils have as much as 10,000 to 20,000 pounds of nitrogen per acre.[55] A particularly interesting case in Texas was described by Kreitler.[56] In addition to natural substances, such as nitrate, many man-related contaminants are stored for years in the unsaturated zone and their presence may lead to continual recontamination of aquifers, as was the case adjacent to the Olentangy River in Ohio.

The long-term effects of oil-field brine holding ponds in central Ohio were investigated for several years by geology students at The Ohio State University. One pond adjacent to a producing well was excavated in 1968. Two years later, when the well was plugged, the holding pond was filled, graded, and seeded. Figure 147 shows that the chloride concentration in the ground a few inches below the water table in the vicinity of the former pond was 36,000 mg/l some 10 years after operation began and 8 years after pond reclamation. This particular area is characterized by a thick and very dense glacial till of low primary permeability.

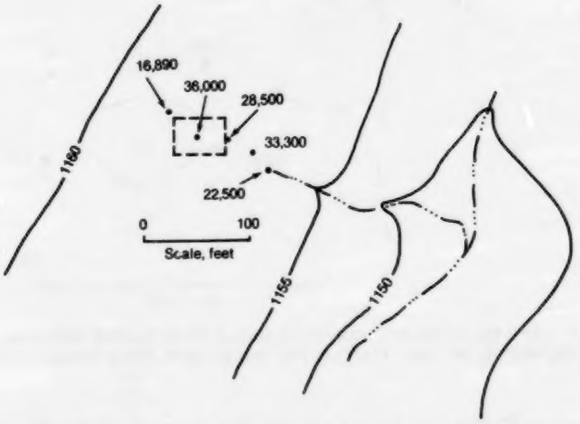


Figure 147. Chloride Concentrations in Shallow Ground Water 8 Years After Reclamation of an Oil-Field Brine Holding Pond

A few miles to the north, three ponds were constructed. Apparently they had not been used for many months when the area was investigated since the chloride concentration in them ranged from 100 to 3,200 mg/l. In the vicinity of these ponds, 18 holes were hand augered to the water table where chloride concentrations ranged from 100 to 16,900 mg/l, as shown in Figure 148. Most of the wells contained less than 2,000 mg/l chloride. In contrast to the area which contained the one pond mentioned above, this three-pond site consists of glacial till which contains several thin layers of sand and gravel. The higher permeability presumably accounts for the overall lower chloride concentration.

The preceding examples illustrate that substantial volumes of both naturally occurring and man-related chemical substances are stored in the unsaturated zone. The quantity of water-soluble substances in storage probably increases with decreasing grain size.

It is commonly assumed that ground-water recharge cannot occur until there is a soil-moisture excess, that is until field capacity is exceeded. Increasing evidence clearly shows, however, that the concept of a distinct wetting front or pistonlike displacement flow through the main matrix of the soil is unlikely in a great many

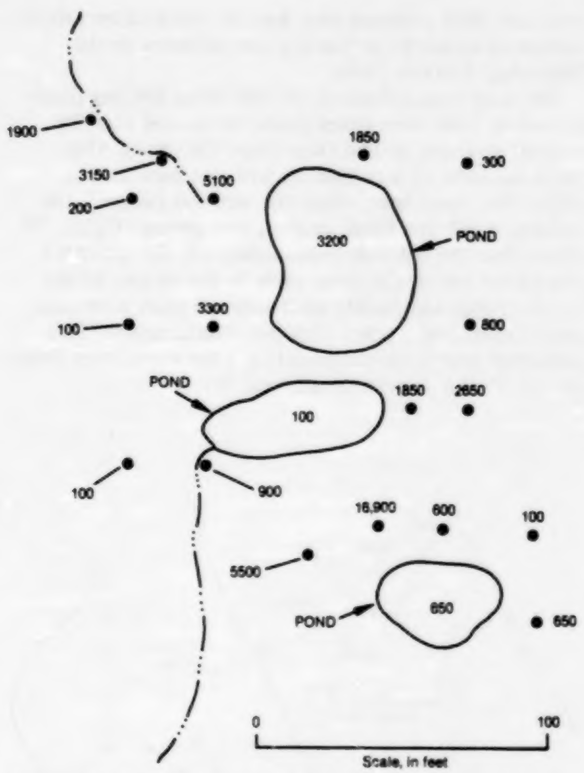


Figure 148. Chloride Concentration in Shallow Ground Water in the Vicinity of Three Inactive Oil-Field Brine Holding Ponds.

cases.[57-67] In fact, in many situations, much of the flow occurs rapidly through macropores and fractures. In a study of the Missouri Ozarks, Aley[68] showed that water entering macropores contributed five times as much recharge as did diffuse flow. Aubertin[69] found that in sloping forested lands, macropores conducted large quantities of water to depths of 30 feet or more. In the majority of cases, at least during parts of each year, ground-water recharge probably is a function both of flow through macropores and fractures as well as displacement flow.

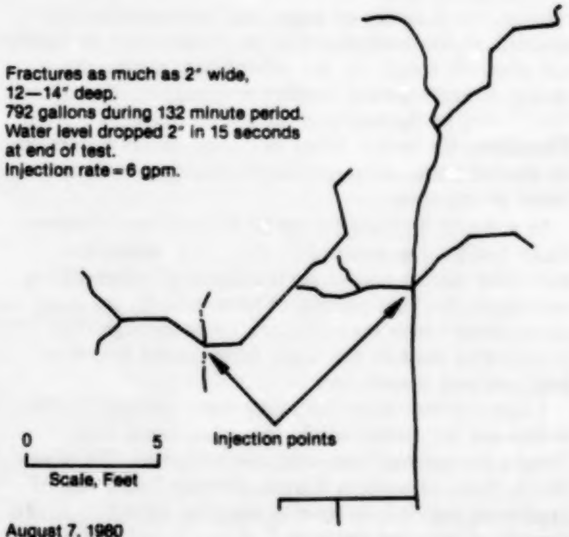
Some macropores and fractures can be exceedingly permeable. While drilling a test hole in north-central North Dakota with a rotary drilling rig, circulation was completely lost at a depth of about 20 feet. Even after mixing two bags of bentonite and a bag of bran in the drilling fluid, it was still not possible to regain circulation in a fine sand. At this time a curious circumstance was noticed—about 30 feet from the rig a fountain of muddy water was flowing from the trunk of an oak tree that had been cut down years before. Apparently, the drill bit had cut through the remains of a large decayed root of the dead tree.

In another case in the same general area, circulation was lost at a depth of about 60 feet while drilling through a thick section of glacial till. After mixing three bags of bentonite with the drilling fluid, circulation was finally regained but not before a spring had formed about 20 feet from the rig. In this case, the drill bit encountered a fracture at some unknown depth that was more permeable than the cutting- and mud-filled annulus of the hole.

The summer of 1980 in north-eastern Oklahoma was exceedingly hot and dry. Large, extensive desiccation crack systems were common throughout the finer textured soil areas within the state, including the author's backyard. Figure 149 shows one of several such fracture systems which was measured on August 7, 1980. This fracture system was about 72 feet long, was as much as 2 inches wide, and ranged from 12 to 14 inches deep. A metal probe could be pushed into the crack to a depth of 34 inches.

Two garden hoses, which provided a combined flow of 6 gpm, were placed in two separate locations in the fracture system. Water flowed into the fracture for 132 minutes (792 gallons). At no time did water overflow the fracture. When the water supply was shut off, the water level dropped 2 inches in 15 seconds. This primitive test clearly illustrates that fracture systems can collect, store and transmit large volumes of water. Furthermore, where extensive desiccation cracks exist, it would be unlikely that much, if any, overland flow could occur except, perhaps, in response to a heavy precipitation event from a convective storm.

Neither macropores nor fractures are commonly as large as those described above. Most may be barely detectable without a close examination. Ritchie and others[60] suggested that the interfaces between adjacent soil peds also serve as macropores. Moreover, these openings need not extend to the land surface in order for flow to occur in them.[66] Nonetheless, water can flow below the root zone in a matter of minutes.



Fractures as much as 2" wide,
12—14" deep.
792 gallons during 132 minute period.
Water level dropped 2" in 15 seconds
at end of test.
Injection rate = 6 gpm.

August 7, 1980
Stillwater, OK

Figure 149. Sketch Map of a Desiccation Fracture in North-eastern Oklahoma

Thomas and Phillips[57] suggested that this type of flow does not appear to last more than a few minutes or perhaps, in unusual cases, more than a few hours after "cessation of irrigation or rain additions".

The preceding discussion demonstrates that the unsaturated zone can store a large volume of water-soluble substances and that macropores and fractures can serve as highly permeable connecting routes between the land surface and the water table. The next consideration is the movement of salts within and from the unsaturated zone to the water table.

When a soil dries, desiccation cracks may form. Moisture on fracture walls may quickly evaporate and a strong capillary potential is developed that draws water from the adjacent soil matrix to the fracture walls. The soil water may be rather highly mineralized, either with natural substances, such as nitrate, or with contaminants that may have been spilled or disposed of on the land surface. Once the soil water reaches fracture or macropore walls and then evaporates, the salts remain as a water soluble lining.

During a period of rain or irrigation, Figure 150 shows that water may flow into these openings, dissolve the water-soluble lining, and rapidly flow downward. This may result in a highly concentrated solution that quickly reaches the water table and substantially degrades ground-water quality. Of course, some of the fluids and salts migrate back into the soil matrix because of head differences and the now reversed capillary potential.

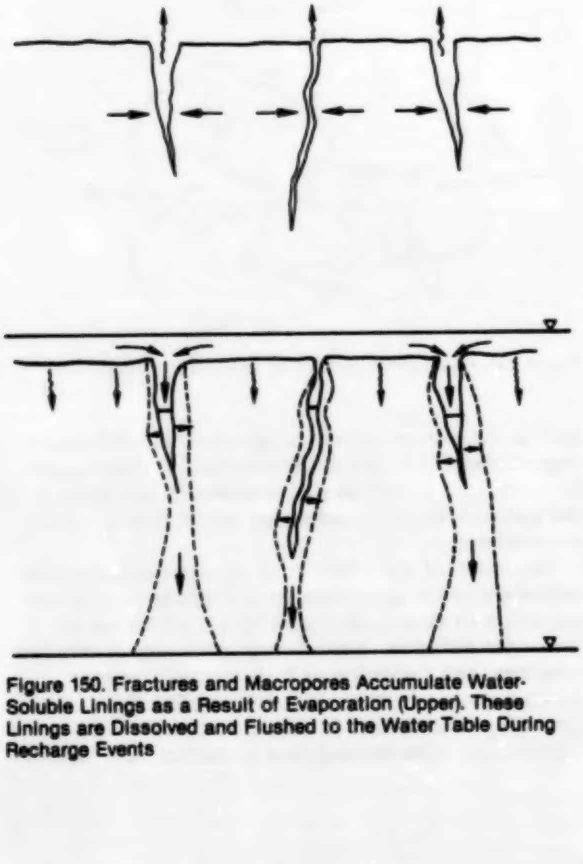


Figure 150. Fractures and Macropores Accumulate Water-Soluble Linings as a Result of Evaporation (Upper). These Linings are Dissolved and Flushed to the Water Table During Recharge Events

Even though there may be a considerable influx of contaminants through macropores and fractures to the water table following a rain, the concentration of solutes in the main soil matrix may change little, if at all. This is clearly indicated in studies by Shuford and others[67] and again shows the major role of large openings. On the other hand, in the spring, when the soil-moisture content exceeds field capacity, some of the relatively immobile or stagnant soil water may percolate to the water table transporting salts with it. A similar widespread recharge period may occur in some places during the fall as a result of decreasing temperature and evapotranspiration and of wet periods that might raise the moisture content above field capacity.

In summary, the preceding suggests the following. During the spring, a large quantity of water-soluble substances may be leached from storage in the unsaturated zone over a wide area. The substances eventually reach the water table and cause significant change in ground-water quality. Although the quantity of leached substances is larger than at any other time during the year, the change may occur more slowly and the resulting concentration in ground water may not be at a maximum because of the diluting effect brought about by the major influx of water over a wide area. A similar but less dramatic change in quality may occur in the fall.

During summer months, ground-water quality changes would be expected to occur more rapidly, perhaps in a matter of hours, because of the large size and abundance of the macropores and fractures. These changes, however, may occur only over a relatively small area because of the local nature of convective storms.

Conclusions

Ecologic conditions in fractures and macropores should be quite different from those in the main soil matrix largely because of the greater abundance of oxygen. Resulting, one might well expect different microbial populations (types and numbers) and chemical conditions in macropores and fractures than in the soil matrix. Coupled with their far greater fracture permeability, this may help to explain why some biodegradable organic compounds or those that should be strongly sorbed actually may reach the water table and move with the ground water. This environment cannot be adequately examined by means of column studies.

Some public health investigators have reported[70] that waterborne diseases seem to increase in the spring and fall. This might conform to and be the result of ground-water recharge through macropores and fractures.

In order to detect and evaluate cyclic fluctuations in ground-water quality and determine background concentrations as well, it will be necessary to install monitoring wells that can be used to measure vertical head differences and collect water samples from discreet sections of the aquifer. Moreover, it will be necessary to collect data frequently, perhaps weekly or even daily, until a pattern can be established.

Prediction of Contaminant Migration

In any ground-water pollution study it is essential to obtain the background concentration of a wide variety of chemical constituents, particularly those that might be common both to the local ground water and a contaminant. As mentioned previously, the water in shallow or surficial aquifers can undergo substantial fluctuations in chemical quality. Therefore, it is not always a simple task to determine background concentrations, particularly of the more conservative constituents, such as chloride or nitrate. In general, samples should be collected during dry periods and not during or within a week following a period of rain. Throughout much of North America the major period of ground-water recharge occurs in wetter periods of the year (generally in the spring), while minor recharge events occur during or immediately after a rain. These recharge events may flush water-soluble compounds from the unsaturated zone to the water table and may substantially change the chemical quality of the ground water. Since the quality of shallow ground water may fluctuate within fairly wide limits during short intervals, it is essential to determine background concentrations statistically by collecting several samples at different times and from different depths.

The severity of ground-water pollution is partly dependent on the characteristics of the waste or leachate, that is, its volume, composition, concentration of the various constituents, time rate of release of the contaminant, the size of the area from which the contaminants are derived, and the density of the leachate, among others. Data describing these parameters are difficult to obtain and are lumped together into the term "mass flow rate", which is the product of the contaminant concentration and its volume and recharge rate, or leakage rate.

Once a leachate is formed it begins to migrate slowly downward through the unsaturated zone where several physical, chemical, and biological forces act upon it. Eventually, however, the leachate may reach saturated strata where it will then flow primarily in a horizontal direction as defined by the hydraulic gradient. From this point on, the leachate will become diluted due to a number of phenomena, including filtration, sorption, chemical processes, microbial degradation, dispersion, time, and distance of travel. Figure 151 illustrates some of these phenomena.

Filtration removes suspended particles from the water mass, including particles of iron and manganese or other precipitates that may have been formed by chemical reaction. Dilution by sorption of chemical compounds is caused largely by clays, metal oxides and hydroxides, and organic matter, all of which function as sorptive material. The amount of sorption depends on the type of pollutant and the physical and chemical properties of the solution and the subsurface material.

Chemical processes are important when precipitation occurs as a result of excess quantities of ions in solution. Chemical processes also include volatization as well as radioactive decay. In many situations, particularly in the case of organic compounds, microbiological degradation effects are not well known. It does appear, however, that a great deal of degradation can occur if the system is not overloaded and appropriate nutrients are available.

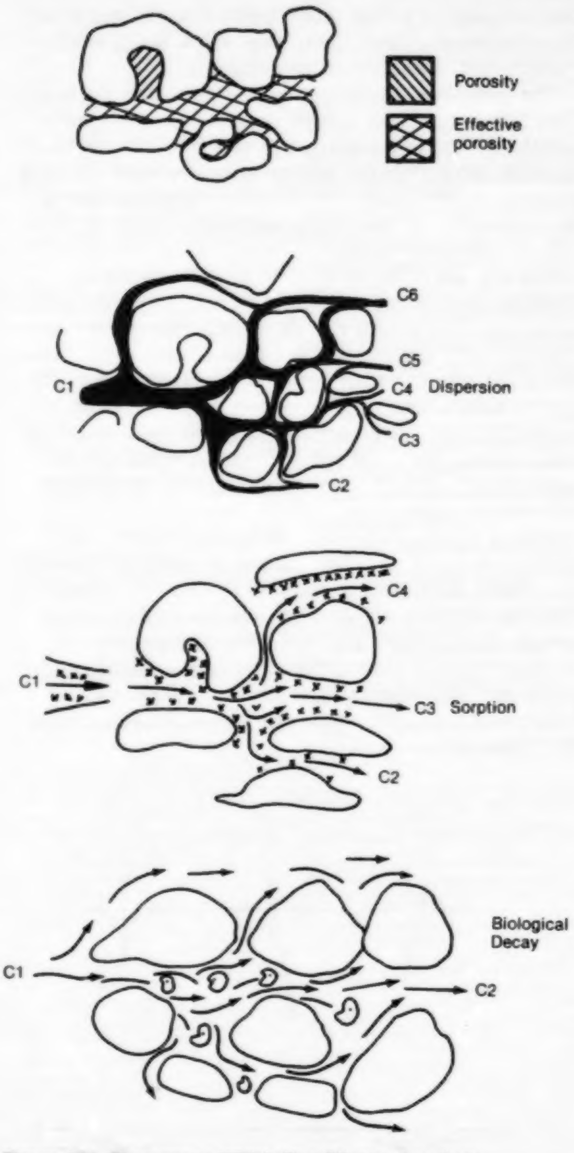
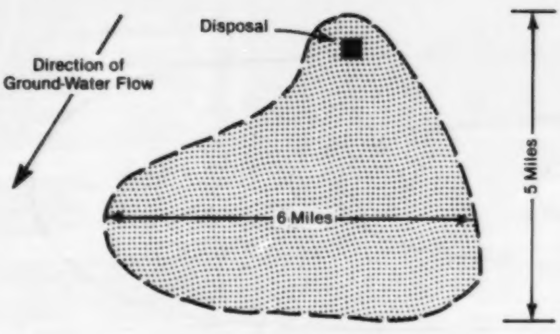
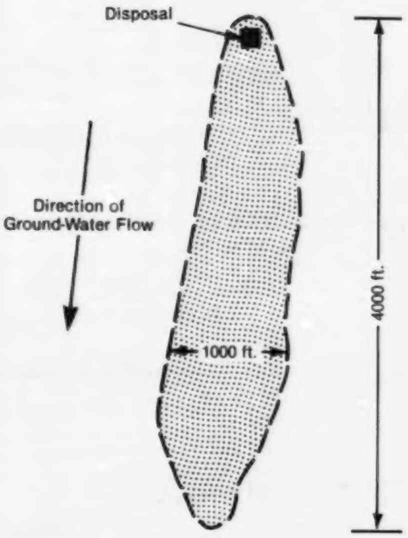


Figure 151. Phenomena Which Can Dilute a Leachate

Dispersion of a leachate in an aquifer causes the concentration of the contaminants to decrease with increasing length of flow. It is caused by a combination of molecular diffusion, which is important only at very low velocities, and dispersion or hydrodynamic mixing, which occurs at higher velocities in laminar flow through porous media. In porous media, different macroscopic velocities and flow paths that have various



a) Chloride Plume, Inel, Idaho
Transverse dispersivity; 450 feet
Time: 16 years



b) Chromium Plume, Long Island
Transverse dispersivity; 14 feet
Time: 13 years

Figure 152. Effect of Differences in Transverse Dispersivity on Shapes of Contamination Plumes.

lengths are to be expected. Leachate moving along a shorter flow path or at a higher velocity would arrive at an end point sooner than that part following a longer path or a lower velocity, which results in hydrodynamic dispersion. Dispersion can be both longitudinal and transverse and the net result is a conic form downstream from a continuous pollution source. As Figure 153 shows, the concentration of the leachate is less at the margins of the cone and increases toward the source. Because dispersion is directly related to ground-water velocity, a plume or slug will tend to increase in size with more rapid flow within the same period of time.

Since dispersion is affected by velocity and the configuration of the aquifer's pore spaces, coefficients must be determined experimentally or empirically for a given aquifer. There is considerable confusion regarding the quantification of the dispersion coefficient and many of the published values are fitted values that cannot be transferred.

Selection of dispersion coefficients that adequately reflect conditions that exist in an aquifer is a problem that can not be readily solved and herein lies one of the major stumbling blocks of chemical transport models.

Often confused with the term dispersion (D_x = longitudinal dispersion and D_y = transverse dispersion) is dispersivity (α_x, α_y). Dispersion includes velocity: to transform from one to another requires either division or multiplication by velocity.

The rate of advance of a contaminant plume can be retarded if there is a reaction between its components and ground-water constitutents or if sorption occurs. This is called retardation (R_d). The plume in which sorption and chemical reactions occur generally will expand more slowly and the concentration will be lower than the plume of an equivalent nonreactive leachate.

Hydrodynamic dispersion affects all solutes equally while sorption and chemical reactions can affect various constituents at different rates. As Figure 154 shows, a leachate source that contains a number of different solutes can have several solutes moving at different rates due to the attenuation processes.

The areal extent of plumes may range within rather wide extremes depending on the local geologic conditions, influences on the hydraulic gradient, such as pumping, ground-water velocity, and changes in the time rate of release of contaminants.

The many complex factors that control the movement of leachate and the overall behavior of contaminant plumes are difficult to assess because the final effect represents several factors integrated collectively. Likewise, concentrations for each constituent in a complex waste are difficult to obtain. Therefore, predictions of concentration and plume geometry, at best, can only be used as estimates, principally to identify whether or not a plume might develop at a site and, if so, to what extent. Models can also be used as an aid in determining potential locations for monitoring wells and to test various renovation or restoration schemes.

A graphical solution (nomograph) was developed to provide a simple computational tool for the prediction of leachate plume movement and corresponding concentration.[71] It is often necessary to estimate the potential distance of travel or length of time required for a plume to migrate some distance in the saturated zone from a point directly below a contaminant source. The concentration of conservative elements in the plume, such as chloride, can be estimated for some selected point in space and time along a flow path that extends directly downgradient from the source.

The nomograph is intended as a rapid means for obtaining an approximate solution. It also aids in understanding the model. It is one-dimensional (restricted to a line).

The Wilson-Miller[72] equation was reformulated to

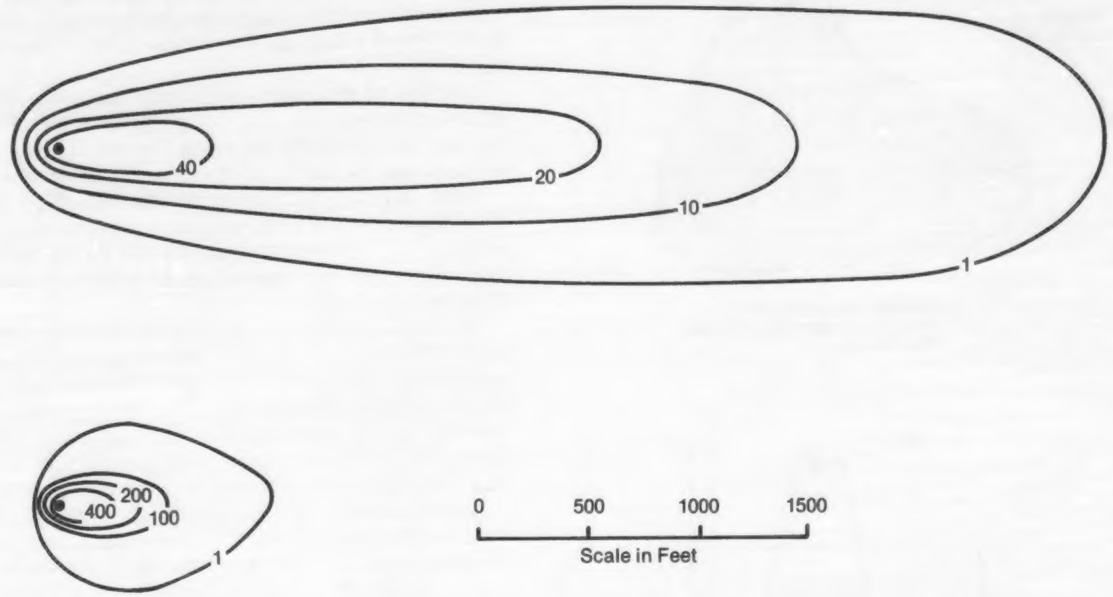
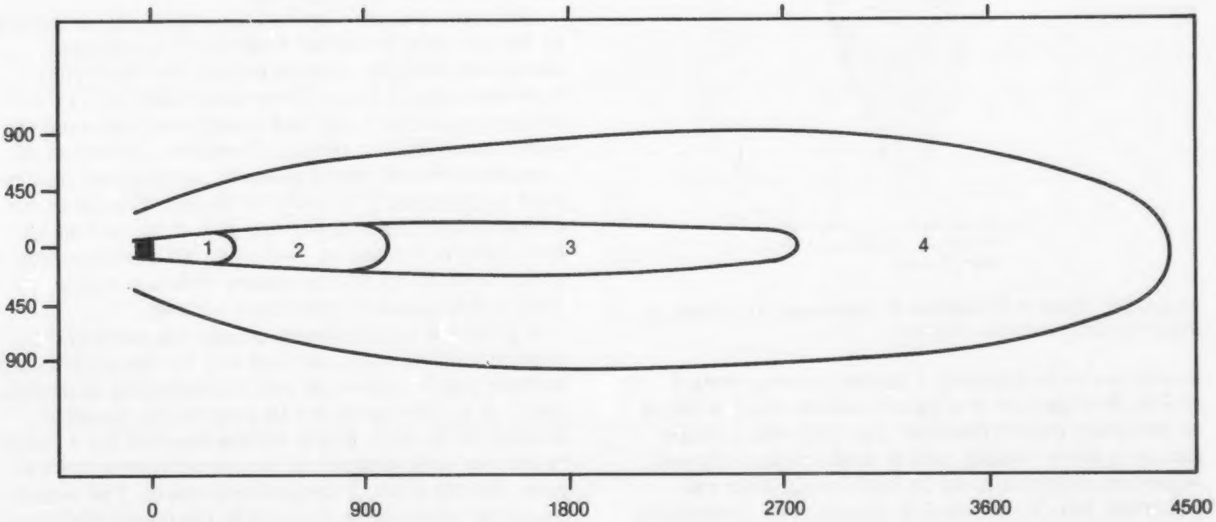


Figure 153. Ground-Water Velocity Exerts a Major Control on Plume Shape. Upper Plume V = 1.5 ft/day and Lower Plume V = .5 ft/day.



1. Chlorobenzene R_d = 35
2. Unknown = 15
3. Chloroform = 3
4. Chloride = 1

Plumes after 2800 days

Figure 154. Plume of a Leachate Containing Several Different Solutes

introduce scale factors and to provide the basis for the nomograph as follows:

$$C = \frac{QC_o}{4\,Q_D\sqrt{\pi x/X_D}} \exp\left[(x/X_D\,\sqrt{\gamma - x/X_D}/2)\operatorname{erfc}(\phi)\right] \qquad (56)$$

where:

$$\phi = \frac{x/X_D - t/T_D}{2\qquad\sqrt{t/T_D}} \qquad (57)$$

Application allows mapping the center-line concentrations of the plume, with respect to time, in one direction (x distance) that is directly downgradient from the source. Dilution-dispersive mixing and retardation parameters are included in the solution. The equation and the nomograph apply to only one chemical constituent, such as chloride or dissolved solids, at a time.

Three scale factors are used in the nomograph as ratios with the primary variables x (distance), t (time), and QC_o, (mass flow rate from source) in the forms of x/X_D, t/T_D, and QC_o/Q_D. The y distance is set to zero. The scale factors are:

$$X_D = \frac{D_x}{V} \qquad (58)$$

$$T_D = \frac{R_d D_x}{V^2} \qquad (59)$$

$$Q_D = nm\,\sqrt{D_x D_y} \qquad (60)$$

Two of the three ratios are computed directly and the third is then found using the nomograph. The factors provide two conveniences. First, the ratios are dimensionless except for concentration. Second, the scale factors combine the constant parameters, which makes it easier to repeat computations of concentration (C) for various positions along the x axis (x) or for different times (t).

The nomograph in Figure 155 is designed to provide a simple technique to estimate one of the following:

Application 1: The concentration (C) at a selected distance (x) and time (t).

Application 2: The distance (x) where a selected concentration (C) will occur at a given time (t).

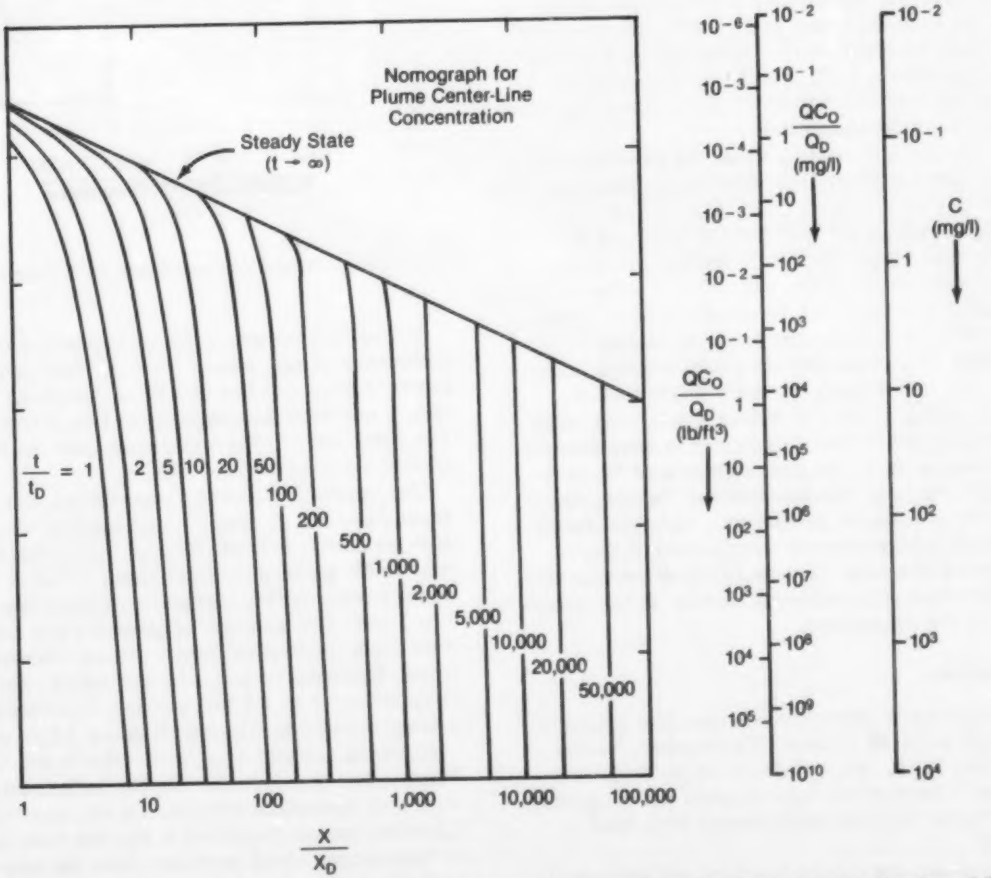


Figure 155. Nomograph for Solutions of Time, Distance, and Concentration for Any Point Along the Principal Direction of Ground-Water Flow.

Application 3:	The time (t) when the concentration will reach a selected concentration (C) at a predetermined location.

As time passes, the concentration in a given area approaches a constant pattern known as steady state. Results for steady-state conditions can be determined for the first two applications as follows:

Application 1b:	The maximum concentration (C) that would occur at a selected distance after a long time period.
Application 2b:	The maximum distance (x) at which a selected concentration (C) will occur after a long time period.

The advantage of applications 1b and 2b is that it is possible to predict, for example, the maximum distance of plume migration for a given concentration threshold or limit. Such concentrations might be those that have been established as standards for the safe drinking water by EPA. Alternatively, it is also possible to predict the maximum leachate concentration that could be reached at a specified distance from the landfill or lagoon.

The estimate of distance (x), time (t), and concentration (C) may require an adjustment of the concentration value to correct for significant background concentrations. In applications 1 and 1b the estimated concentration (C) must be added to the background concentration. In applications 2, 2b, and 3 the concentration value used must be the remainder after subtracting the natural background concentration.

The nomograph provides a visual representation of the plume concentration. The solution can be found easily for various locations, times, and concentrations. This leads to gaining a "feel" for the nature of the plume. As time passes, the concentration at a given location reaches steady state. The steady state value for concentration can be useful, for example, as a "worst case" scenario (maximum concentration reached in the infinite time). The upper line on the nomograph represents the time and distance at which steady state is reached. It is easy to see that before steady state, small changes in location or time correspond to large changes in concentration. In a sense, the steepness of the non-steady state time lines indicates that the "leading edge" of the plume is relatively narrow and, therefore, passes a given location in a relatively short period of time. Behind the leading edge, the concentration remains constant at the steady-state value represented by the steady-state line on the nomograph.

Example Problem

The ground-water contamination case that is used in this example occurred in South Farmingdale, Nassau County, New York, and was described by Perlmutter and Lieber.[73] Most of the data required for the solution described below were obtained directly from their report.

Contamination was caused by cadmium- and hexavalent chromium-enriched electroplating wastes that infiltrated from disposal basins into a shallow glacial aquifer. Apparently disposal began in 1941 and continued intermittently for several years. By the early 1960's a leachate plume originating at the disposal ponds extended downgradient about 4,300 feet, and was as much as 1,000 feet wide and as much as 70 feet thick. Figure 156 shows that the plume extends to the headwaters of Massapequa Creek, a small stream that serves as a natural drain for part of the contaminanted water.

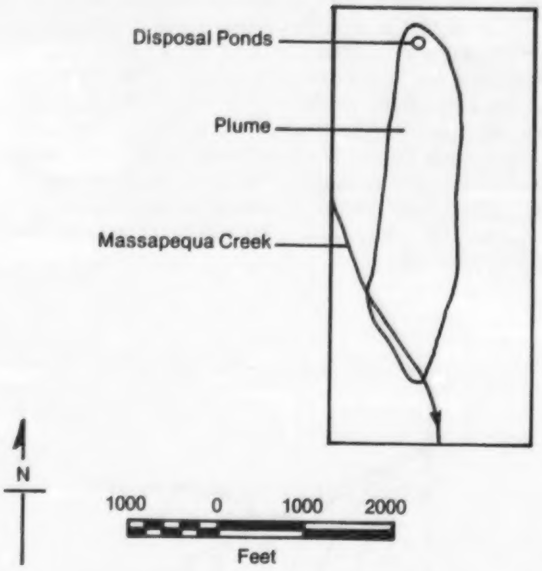


Figure 156. Leachate Plume at South Farmingdale, New York

The surficial or upper glacial deposits, which are Pleistocene in age, extend from the land surface to a depth of 80 to 140 feet and lie on the Magothy Formation, a unit of stream deposits of Late Cretaceous age. The water table in the surficial deposits lies from 0 to about 15 feet below land surface.

The aquifer is in dynamic equilibrium and receives about 22 inches or about 1 mgd (million gallons per day) per square mile of recharge from precipitation. The water-table gradient averages about 1 foot in 500 feet and the water table undergoes an annual fluctuation of 2 to 3 feet. The direction of ground-water flow is southward from the disposal ponds toward Massapequa Creek. Estimates of ground-water velocity for the area range from 0.5 to 1.5 feet per day. Reportedly, the average velocity for the area is about 1 foot per day.

Chemical analyses of ground water in the South Farmingdale area indicate that the background concentration of hexavalent chromium is less than 0.01 mg/l. Likewise, the concentration is also less than 0.01 mg/l in Massapequa Creek upstream from the area of the leachate plume. Along that stretch where the plume discharges into the stream, the concentration of chromium is substantially greater.

As much as 200,000 to 300,000 gallons per day of effluent (equivalent to 52 pounds per day of chromium) were discharged during the early 1940's to three disposal pits, which have a combined area of about 15,470 square feet. Since 1945 the volume of the waste stream has been reduced substantially and eventually a treatment plant was constructed. On two occasions the chromium concentration in the raw effluent was 28 and 29 mg/l.

The relatively clean nature (free of clay or organic matter) of the materials forming the surficial aquifer precluded any significant reduction in the chromium load in the plume. That is, ion-exchange during movement was negligible. Maximum chromium concentrations in the plume ranged from about 40 mg/l in 1949 to about 10 mg/l in 1962.

The plume is about 200 feet wide at its origin at the disposal ponds. It reaches a maximum length of about 4,300 feet and increases in width to about 1,000 feet. Assuming a velocity of 1.5 feet per day, this plume has an average longitudinal dispersion (D_x) of 105 ft^2/day, a transverse dispersion (D_y) of 21 ft^2/day, and dispersivities of 70 ft (α_x) and 14 ft. (α_y), respectively. Table 13 shows a summary of the required data. Application of the three methods to this example follows.

Table 13. Summary of Data for Example 1

thickness:	m	= 110 feet
porosity:	n	= 0.35
velocity:	V	= 1.5 ft/day
dispersion:	D_x	= 105 ft^2/day
	D_y	= 21 ft^2/day
retardation:	R_d	= 1
volume flow rate:	Q	= 26,800 ft^3/day
source concentration:	C_o	= 31 mg/l
mass flow rate:	QC_o	= 26,800 ft^3/day x 31 mg/l
or	QC_o	= 52 lb/day

Application 1, illustrated in Figure 157:

To find concentration (C) for a distance (x) of 4,200 feet from the source and time (t) of 2,300 days, calculate:

$$\frac{x}{X_D} = \frac{4{,}200 \text{ ft}}{70 \text{ ft}} = 60 \text{ (Locate at A)}$$

$$\frac{t}{T_D} = \frac{2{,}300 \text{ days}}{46.7 \text{ days}} = 50 \text{ (Locate curve E)}$$

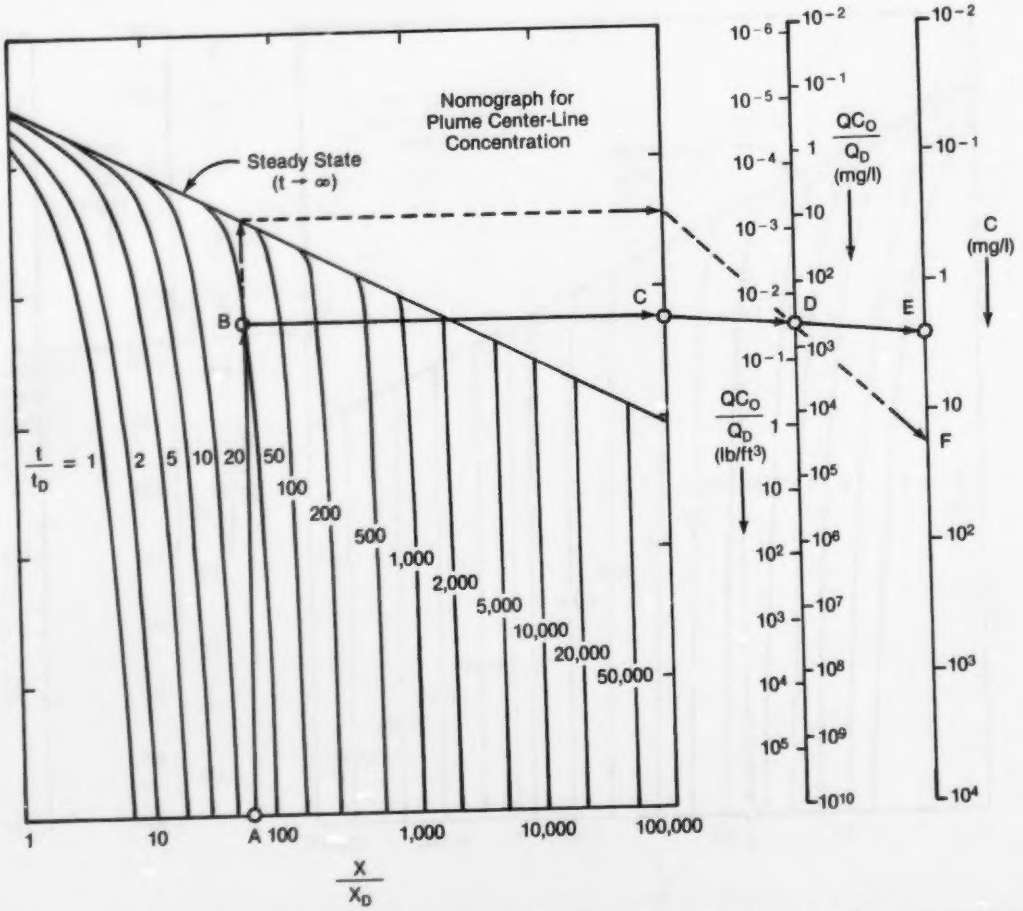


Figure 157. Applications 1a and 1b: Using the Nomograph to Estimate the Concentration Given Values of Distance and Time

$$\frac{QC_o}{Q_D} = \frac{(26{,}800\ \text{ft}^3/\text{day})\,(31\ \text{mg/l})}{1{,}800\ \text{ft}^3/\text{day}}$$

$$= 460\ \text{mg/l (Locate at D)}$$

or:

$$\frac{QC_D}{Q_D} = \frac{52\ \text{lb/day}}{1{,}800\ \text{ft}^3/\text{day}} = .029\ \text{lb/ft}^3\ \text{(Locate at D)}$$

Using Figure 157 draw a line vertically from A to the intersection with the t/T_D curve B, then horizontally from B to C and from C through the scale D to E, giving a concentration (C) of 2.6 mg/l.

Application 1b, illustrated in Figure 157:

To find the maximum concentration for a given distance for large time, use the steady-state line instead of curve (t/T_D). Proceeding as above, a concentration (C) of 20 mg/l is read at F.

Application 2, illustrated in Figure 158:

To find distance (x) where a concentration (C) of 2.6 mg/l will occur at a time (t) of 2,300 days, calculate:

$$\frac{t}{T_D} = \frac{2{,}300\ \text{days}}{46.7\ \text{days}} = 50\ \text{(Locate at D)}$$

$$\frac{QC_o}{Q_D} = \frac{(26{,}800\ \text{ft}^3/\text{day})\,(31\ \text{mg/l})}{1{,}800\ \text{ft}^3/\text{day}}$$

$$= 460\ \text{mg/l (Locate at B)}$$

Using Figure 158, locate the selected concentration at A. Draw a line from A through B to C, then horizontally from C to curve D, and vertically to E, giving:

$$\frac{x}{X_D} = 60$$

Multiply by X_D to determine distance (x):

$$x = \left(\frac{x}{X_D}\right)(X_D) = (60)(70\ \text{ft}) = 4{,}200\ \text{feet}$$

Application 2b, illustrated in Figure 158:

To find the maximum distance at which a selected concentration will reach a given value for large time, use the steady-state line instead of curve (t/T_D) in Figure 158.

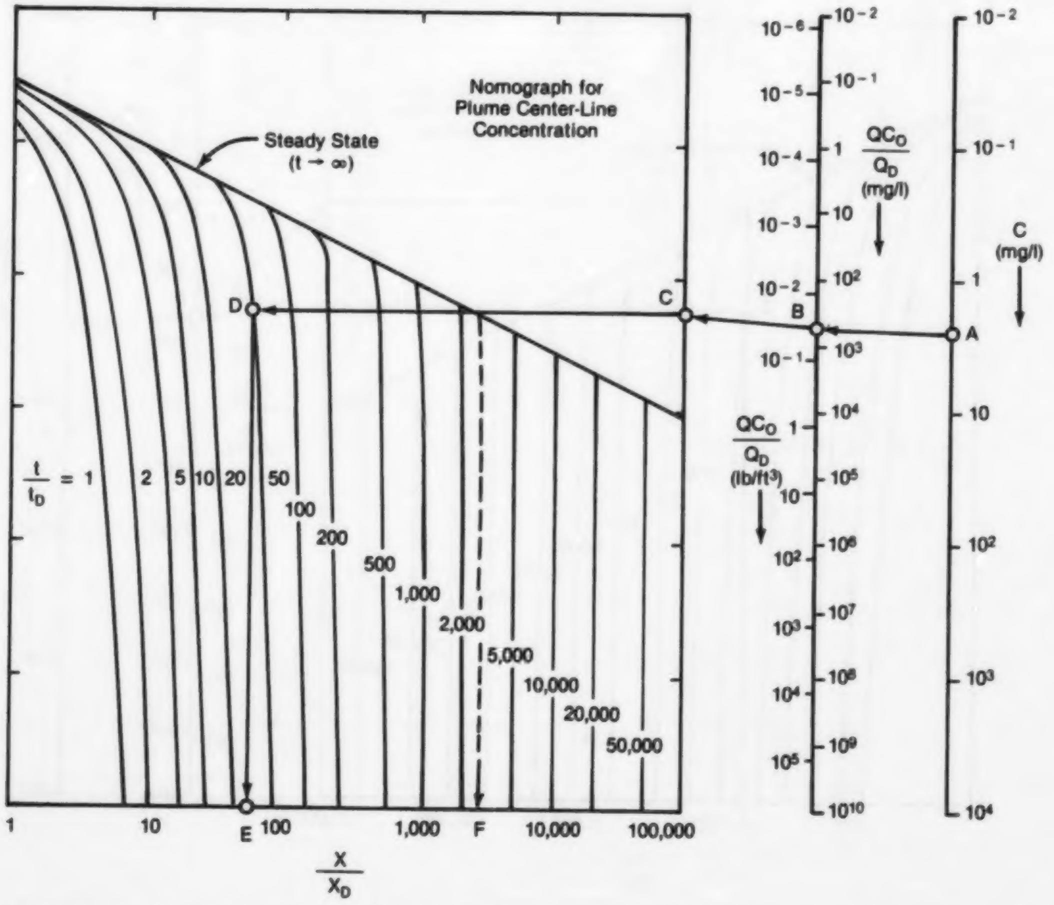


Figure 158. Application 2a: Using the Nomograph to Estimate the Distance for Given Values of Concentration and Time

Application 3, illustrated in Figure 159:

To find time (t) when the concentration (C) will reach 2.6 mg/l at location (x) of 4,200 feet, calculate:

$$\frac{x}{X_D} = \frac{4{,}200 \text{ ft}}{70 \text{ ft}} = 60 \text{ (Locate at D)}$$

$$\frac{QC_o}{Q_D} = \frac{(26{,}800 \text{ ft}^3\text{/day})(31 \text{ mg/l})}{1{,}800 \text{ ft}^3\text{/day}}$$

$$= 460 \text{ mg/l (Locate at B)}$$

Using Figure 159, locate the selected concentration at A, draw a line from A through B to C, then horizontally from C to an intersection with a vertical line from D, giving at E:

$$\frac{t}{T_D} = 50$$

Multiply by T_D, to determine time (t):

$$t = \left(\frac{t}{T_D}\right)(T_D) = (50)(46.7 \text{ days}) = 2{,}300 \text{ days}$$

If the lines intersect above the steady-state line, the concentration will not reach the given value at that location.

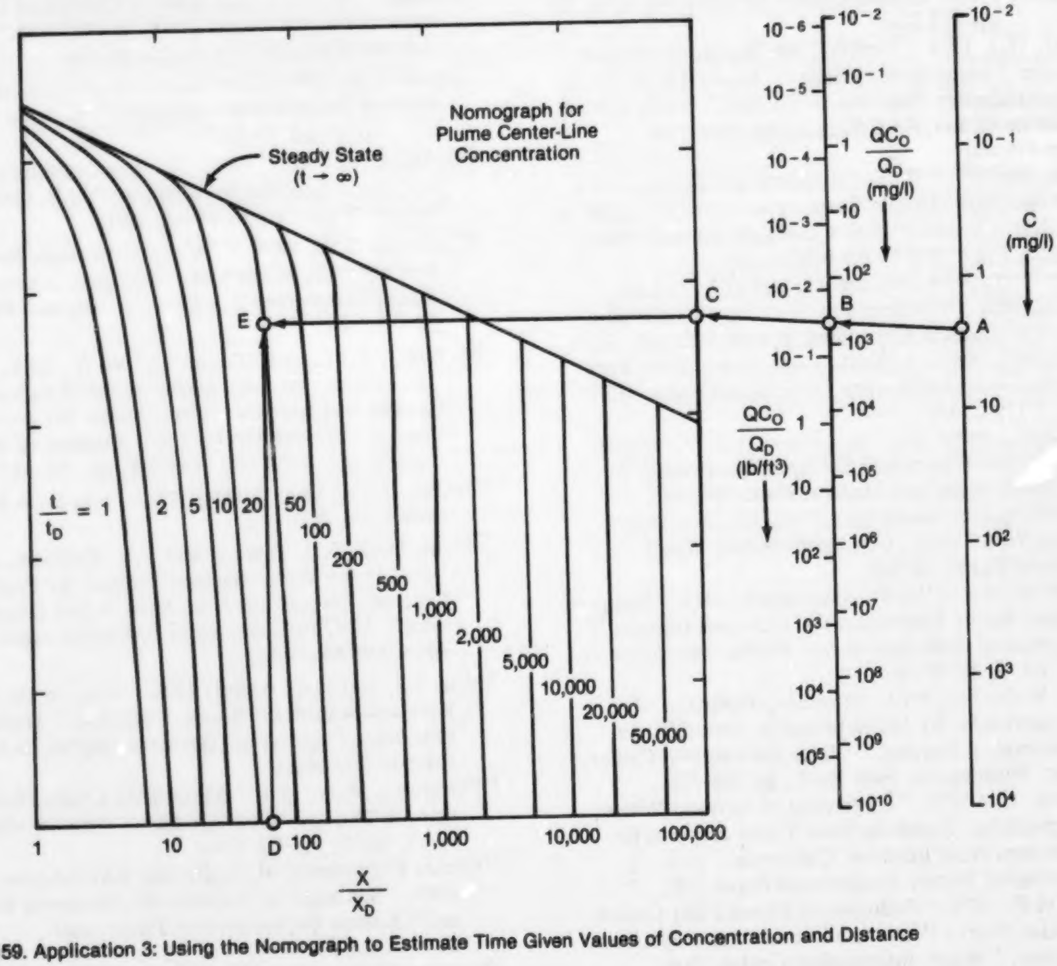


Figure 159. Application 3: Using the Nomograph to Estimate Time Given Values of Concentration and Distance

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Chapter 6

Management Alternatives

Introduction

All of the available case studies of ground-water contamination were reviewed to collect the various strategies used by local public water supply managers. These case studies showed that the Water Utility Manager is a very small player in major contamination incidents. Often he controls few of the resources available and the drinking water solution is considered a low priority by the various agencies involved. From the local manager's point of view, the immediate problem is providing safe water to the public in sufficient quantity and at a price the community can afford. Other agencies and interests may have other objectives. What is the bottom line? For the manager there are two main facts: (1) prevention is probably all a community can afford and (2) safe drinking water presents a cluster of problems, issues, and objectives which must be addressed by the whole community.

Hindsight and the Rubick's Cube

Although you have no current problem, you should be doing a great deal about ground-water contamination. The easiest way to understand why is to see what has happened to other communities and to ask a number of "what if" questions. Unfortunately, most people have learned what they should have done to prepare and prevent contamination by discovering contaminated wells.

EDB Found in Well Water At Southside Va. School*

By Tom Sherwood
Washington Post Staff Writer

RICHMOND, Feb. 10—*Gov.* Charles S. Robb said today that the suspected cancer-causing agent ethylene dibromide (*EDB*) has been found in the well water of a rural elementary *school* in Southside Virginia's Halifax County.

The 215-pupil school, located in the farming community of Turbeville near the North Carolina border, immediately *suspended* most uses of the contaminated well pending more tests by state officials and the federal Environmental Protection Agency, according to *acting School Superintendent* S. Dail Yeatts.

The EDB announcement was the second action taken in Virginia since the EPA announced an emergency ban on agricultural uses of the pesticide chemical almost two weeks ago. Last week, Virginia officials announced that two prepared muffin mixes were being removed from grocery shelves.

An *EPA spokesman* in Washington said his agency had not yet received a report on the well water, but said initial indications are that the Virginia school contamination "is *relatively low*." He added: "*But we're concerned about any level of EDB contamination.*"

Two state test samples showed a contamination level of .027 parts per billion of water, Robb's office said. The minimum detection level for EDB is .02 parts per billion.

The school's well was the only one that showed EDB contamination in tests of 87 public water supply wells in 17 counties and one city of Southside and southeastern Virginia, Robb's office said.

Those heavily farmed areas were identified by the Virginia Department of Agriculture as the most likely to be contaminated by EDB.

School Superintendent Yeatts said in a telephone interview that Halifax officials were informed of the contamination late today. He said a freshwater tank was sent to the school for drinking.

"We're taking *portable tanks of water* to wash dishes in the cafeteria. We're going to use *paper plates* and plastic utensils," Yeatts said. "We're doing everything we can to protect the children."

Yeatts said officials were uncertain when the well was established, but said it was the only source of drinking water for the school.

He said the school, built in *1932*, is located in an isolated area about 20 miles from the town of Danville. It is surrounded by farmland that produces tobacco, grain, and canteloupes, a major crop in the community that sponsors an annual canteloupe festival in the spring.

Robb's office said long-term solutions to the well problem included the possibility of *drilling another well* or adding a *carbon filtration system*.

A state task force is continuing its test of grain products, officials said.

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The case studies we have read and the ones we have heard contain many lessons. The lessons can all be boiled down to show that responding to contaminated public wells is not a technical problem *alone*. It *is* a technical problem, requiring professional knowledge of hydrogeology, engineering, and other disciplines; but it is also a financial problem, a legal problem, a public affairs and public information problem, a health effects and risk assessment problem, and in the final analysis, fundamentally a community problem because only the whole community can decide what the future of the community should be. This is why the problem is like a Rubick's cube; you must coordinate all sides of the cube.

When you discover contamination or a spill is reported, the questions from your customers and the press do not come in logical order. Everyone wants to know: "Where is the contamination coming from?; Is the water safe to drink?; Have we already been affected?; What are you doing about it?"

A newspaper article, reporting EDB in the well of a small school in rural southern Virginia, is reproduced on the previous page. This article points out the problems which hit the manager of a water supply. The acting superintendent is called by a major newspaper the same day the governor's office announces the results of the sampling. Concurrently there are questions about adequate protection for the children, an alternate water supply, and the health effects. All these areas must be addressed immediately by a local school board and superintendent.

Where do you start? As superintendent Yeatts did, you start with temporary measures to remove health risks and, equally important, to control fear. Whether the children were in any immediate danger or not, their parents and the public needed to be reassured. Then the local community can work out a longer temporary solution, giving them time to decide the future of the school and its water supply.

Perhaps this is an extreme example, but it does seem typical for several reasons. The incident was unexpected and apparently there was no emergency plan. In a case like this, you usually wish that you had taken the time to collect information about your wells and aquifer before the contamination was discovered.

The previous chapters serve to illustrate how often clean water can be found with contamination close by. There are methods to protect an aquifer from an approaching plume. You must take the time to find out which hydrogeologists have studied your aquifer and to collect and evaluate any results of samples taken from private wells. You should put together a good map of your aquifer. You also should calculate the influence of all the cones of depression as the various wells are turned off and on. This would tell you the real gradient and allow you to calculate the travel time of a hypothetical slug to your well. Using these methods, you can collect and analyze a lot of free information about your wells and aquifers and calculate various possible contamination problems. You should also get more than one opinion about the local hydrogeology. Hydrogeologists do not always agree, and amazingly, sometimes are wrong. In some cases, large amounts of money have been spent only to discover that the slurry wall did not work because the aquifer was deeper in places than calculated. So get as many opinions as you can.

Legal Issues

Before you discover that you have contaminated wells and all kinds of questions need answers, you should take the time to look into Federal laws, but more importantly, into your own State law. Can the customers sue? What is your obligation to inform the public if the contaminant is not regulated? Who is responsible for the contamination? Can we prove it? Do we have to sue to get them to clean it up? Do we qualify for Superfund help? You may already know there are cases of utilities being sucessfully sued even when no negligence was involved. What is the situaiton in your State? We also have heard of cases where Superfund came in and provided emergency help and then pulled out because the immediate threat to public health was over.

Financial Issues

You probably have had enough trouble getting the water rates up to a level where you could pay for operation and maintenance. There is no operating reserve. What happens when you discover contamination? What can you afford to do? What does bottled water cost? Can you reduce pumping? Can you hook up to a neighboring utility? Can you get other well owners to cooperate? What will it cost? Will your customers be willing to pay? What about treatment? What do granular activated carbon and airstripping cost? How long will it take to find out what you need? Who can tell you? You must put together some contingency plan to deal with raising money and negotiate the options with the community. Once you discover contamination there's no time to arrange for loans or bonds, and Federal funds seem a very distant possibility. What happens if the health department orders you to take steps you cannot afford? What happens to the community if you go bankrupt?

Public Information and Health Effects

You now have a report from a lab confirming the presence of a contaminant in the water from your wells at a specific concentration. Most likely the contaminant is a man-made chemical, a suspected carcinogen, and present in trace amounts; e.g., parts per billion. What do you tell your customers, reporters, the public?

It's very late, certainly too late to start educating your customers about what "safe" means. One thing is certain, they are not going to be reassured by statements concerning risks of ten to the minus sixth. EPA is trying to come to grips with this problem. The December 1984 *EPA Journal* is devoted to the problems of risk assessment and risk management and the public's role in these areas. EPA recognizes that in the end it is the local community which must decide how to deal with risk management. This is why it is impossible to separate the issue of public information from the issue of risk management.

When referring to contamination, your customers most certainly will tell you, "take it all out." They want no risk at all. Do they understand how much it will cost and the consequences for the community of spending all that money on drinking water? Will it discourage new industry? Will employment suffer? Will people just move away from high water rates? What is the benefit of treating to zero? You can see that it is very late to start asking the citizens such questions when there is publicly reported contamination.

The public's first question when contamination is reported is, "Is it safe to drink?" You have to give them the truth as soon as you have it or any trust you have built will be destroyed. What if you, and the state, and EPA do not honestly know the health effects for sure? Then you have to say so, although as Mr. Ruckelshaus says in his interview in the December *EPA Journal*, people will tend to think we really do know and are not telling them. So what is the problem—educating the public or determining the status of research on health effects? Both are the problem. Once you have contamination you have to start a crash course for the public on the risk involved with the particular chemical at the specific level detected. The public may not be satisfied because they are unaware of how such risks are evaluated, and therefore, will be unsatisfied with your statistical talk and lack of clear affirmative statements. Too bad you did not try to educate them sooner. Now you will have to present them with a plan involving costs, and perhaps allowing certain levels of the contaminants to remain in their water. How clean is clean?

The remainder of this chapter provides some guidelines for developing a technical plan to prevent and to respond to contamination.

Just because a water-quality problem is not known to exist at the present time does not mean that one may not appear tomorrow or at some other time in the future. Contingency plans should be developed before a problem develops so that sufficient time is available for rational decisions and planning. Several approaches can be followed, any of which should be dictated by the particular political, economic, and technical situations that exist. The contingency plans need not be expensive nor should they necessarily follow traditional methods. Some of the best ideas are generated by individuals with little or no scientific training, but these same people are characterized by a great deal of common sense and a need to quickly and inexpensively solve a problem.

Although any management plan must be flexible, a number of steps can be followed that should make the plan easier to implement. Certainly not all inclusive, at least the following steps could be taken: (1) determine where the supply originates and what problems might be associated with it, (2) learn the system, (3) locate potential sources of contamination, (4) develop a system of self monitoring, (5) consider alternate sources of permanent or temporary supply, (6) locate and evaluate existing laws and regulations on waste disposal, (7) develop an aquifer sensitivity model, and (8) develop emergency response plans.

Where Does the Supply Originate

A short distance from its border with Kansas, Oklahoma's Cimarron River contains more than 50,000 mg/1 of dissolved solids during low flow conditions. Scores of miles down stream, near its confluence with the Arkansas River, the Cimarron still contains more than 2,000 mg/1 of dissolved solids despite the dilution from several major tributaries. In this case any wells drilled in the flood plain that are dependent on induced infiltration soon would be contaminated. The source for the calcium sulfate and sodium chloride in the river is natural, being derived from a series of saline springs and seeps.

At Minot, North Dakota, two muncipal wells produce water that contains higher concentrations of chloride than do the other wells in the field. The two wells are also about 50 feet deeper than the average. In this case a buried interglacial river valley trends through the center of town; it had cut several tens of feet into the underlying bedrock. One of the bedrock formations that subscrops along the buried valley walls is the Cannonball Formation, a unit that contains salty water. Apparently the high chloride wells are screened near the subscrop of the Cannonball and when pumped induce salty water to flow from the Cannonball, mix with the fresher water in the glacial sand and gravel, and eventually reach the municipal wells. This problem also is natural and perhaps the most practical control is to blend the water with that from other wells.

Several years ago, in an industrialized city in Michigan, a plant water manager decided to dredge the adjacent river in order to increase the yield from their induced infiltration supply. Not realizing the river contained high concentrations of a variety of industrial wastes, the river was dredged and within days the chemical quality of the well water deteriorated dramatically. It was then recognized that waste papermill products had sealed the river bottom, providing a last line of defense between the contaminated river and the well field.

It is evident from the above that a knowledge of the origin of the water-supply system can serve as a starting point in the development of management alternatives.

Learn the System

For the most part, geologic and hydrologic evaluations of the subsurface are based on an analysis of logs of wells and test holes. Unfortunately, these data commonly are not readily available, although experience has shown that they are likely to be stored in a file some where. If they do not turn up, it might be possible to obtain copies from the original driller, contractor, or consultant. Well construction details, such as depth, length of screen, etc., also can be of considerable value. The geologic data can be used to construct a number of maps and cross section of the aquifer system, as shown in Figure 160. Cross sections should provide an idea of how much protection the aquifer and confining units provide against contamination.

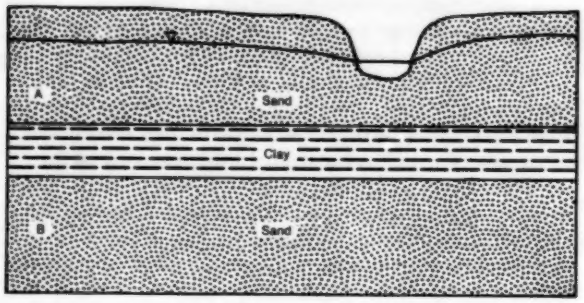

Figure 160. Aquifer A Has No Natural Protection and Is Highly Subject to Contamination from the Surface. Aquifer B is Protected to Some Degree by the Overlying Layer of Clay

Notice in Figure 160 that the shallow or surficial aquifer (A) consists entirely of permeable material that extends from the land surface to the base of the water-bearing unit. Consequently, this aquifer has practically no natural protection other than the thickness of the unsaturated zone. It could be easily contaminated by a spill or nearly any type of waste disposal scheme. On the other hand, the deeper aquifer (B) is covered by a confining unit of low permeability, one that might require years for a contaminant to penetrate. In this case the deeper aquifer has some degree of natural protection and, in the case of a spill, time likely would be available to develop plans to overcome a potential contamination problem.

Well production/acceptance or aquifer test results also can be very useful as aids in understanding how the ground-water system functions. In both production and aquifer tests, water levels are measured in a pumping well and in one or more observation wells during a time period that commonly exceeds eight hours. These data, plotted as drawdown versus time or as drawdown in several wells versus distance from the pumped well, can be used to determine an aquifer's hydraulic conductivity or transmissivity, as well as the storativity. These parameters are necessary to calcuate ground-water velocity, among other things (see Chapter 2).

Nearly as important as geologic information are records of well discharge (rate and time interval) and water-level fluctuations, the latter indicating how the aquifer acts under stress. Of particular concern is the size of the cone of depression around a pumping well. As described in Chapter 2, the radius of the cone of depression is controlled by the aquifer properties and the discharge rate. In an unconfined aquifer, the radius of the cone may be in the order of a few hundred feet, but in a confined aquifer it may extend outward for miles. Furthermore, the drawdown caused by overlapping cones of depression is greater than that caused by a single well. Additionally, horizontal and vertical variations in aquifer properties, pumping schedules and rates, and well interference will tend to distort the shape of the composite cone of depression in a well field.

The shape and areal extent of the cone of depression are important in contamination studies for two major reasons. First, the cone represents a change in the hydraulic gradient, which steepens as it approaches the well. This, in turn, increases the velocity of ground-water flow. Second, contaminants that reach the aquifer and are within a cone will migrate toward the pumping well. Therefore, what ever happens within the radius of influence of a well is of concern to the water manager and this is why a knowledge of the size of the cone of depression is so important.

Likewise, the size of the area of influence of a well field is important because this is the area that should be protected, as shown in Figure 161. Although the area of influence might well exceed several square miles, the velocity of the ground water near the outer margin should be relatively low, as compared to the velocity in the vicinity of a well. If the aquifer were contaminated in this region, it might require several months or even years for the contaminant to appear at a well. In the meantime, the contaminant might be diluted or degraded to such an extent that it would not be of concern to the plant operator.

Locating Potential Sources of Contamination

In order to develop a mangement plan, local sources or potential sources of contamination must be known. These include, in addition to the more obvious ones, such things as the location of railroads, major highways, gasoline stations, and small industrial or service plants, particularly those small concerns that might be operated in someone's garage, basement, or outbuilding. The latter are not likely to be well known to regulatory agencies nor are they likely to have discharge permits.

In order to develop a data base for potential sources of ground-water pollution, several waste surveys could be conducted. These should include: (1) an industrial waste survey, (2) a municipal waste survey, and (3) a state and federal property survey. The surveys could be as simple as examining a map to locate highways, railroads, industrial sites, disposal sites, etc., or as complex as interviewing a large number of people. Surveys of this nature could become both time consuming and expensive, particularly if done inhouse.

One approach, which could be both comprehensive and inexpensive, would be to contact a number of local service clubs, such as the Lion's Club, and request their members to provide information. Other than the obvious advantages of this method, one important consideration is that the members of local service clubs commonly represent a wide spectrum of the population that, as a whole, might well have a detailed knowledge of the area. The data base that could be developed

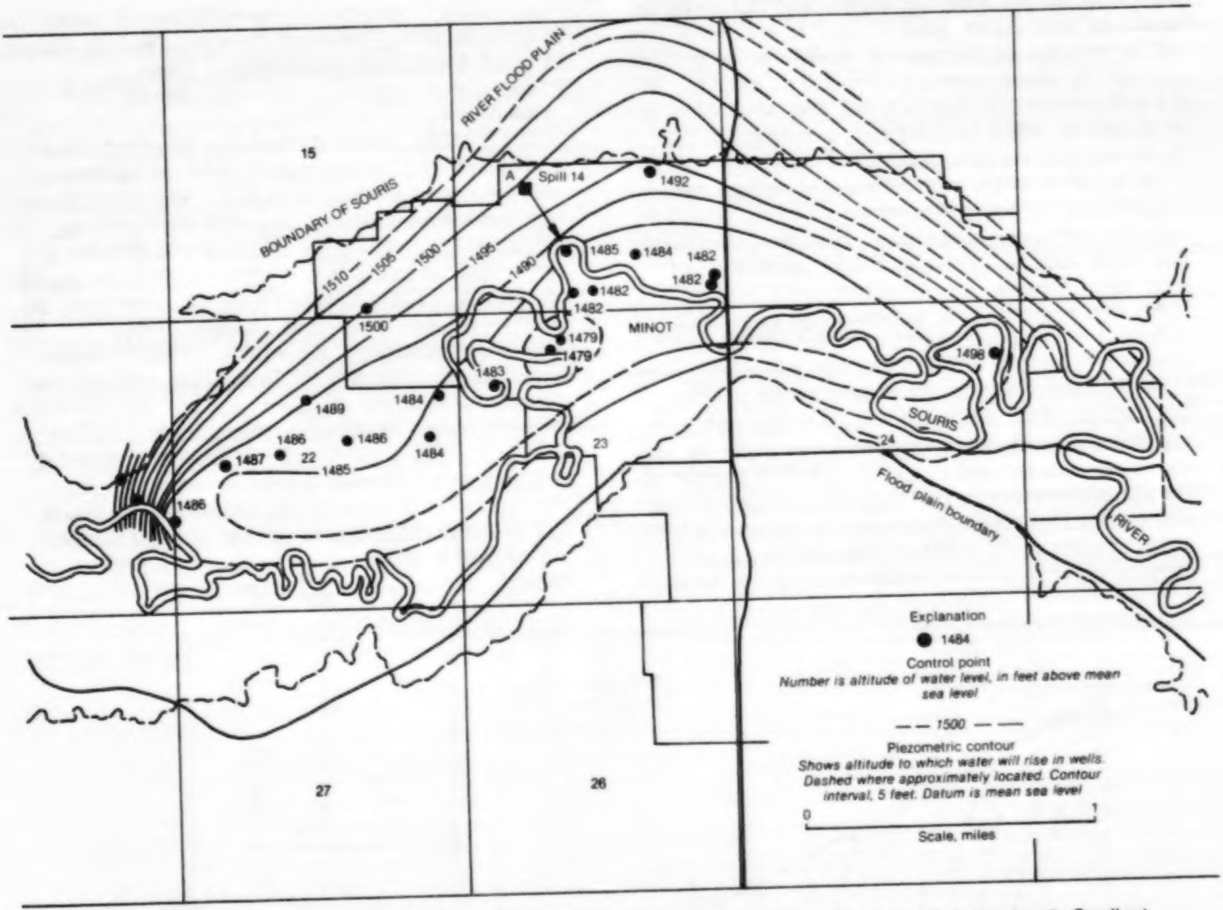

Figure 161. Water-Level Contour Map of The Main Aquifer Showing a Large Cone of Depression and Steep Hydraulic Gradient

would potentially be comprehensive, inexpensive, and flavored with community pride.

Potential Contaminants and Travel Times

Once the location of potential sites for contamination are located, it will be possible to estimate the time required for a contaminant to migrate to a well. It must be remembered, however, that what ever method is used to predict travel time, it will be only an estimate. Three methods that can be used are a simple equation, a nomograph, and a computer model.

As discussed in Chapter 2, ground-water velocity is controlled by the aquifers hydraulic conductivity (K), effective porosity(n), and hydraulic gradient(I), that is,

$$V = KI/7.48n. \qquad (61)$$

The hydraulic gradient is measured along a flow line that originates at the contaminant source and continues to the closest well in the downgradient direction. Of course, the flow line must be drawn so that it intersects the water-level contours at right angles. The easiest way to determine the average gradient is to subtract the water level at the well from the water level at the contamination site and then divide this number by the distance between the two points as measured along the flow line.

Let us assume that a spill occurs at point A as shown in Figure 161. The spill consists of leakage of 2,000 gallons a day for three days of salty waste water that contains 30,000 mg/1 of chloride. The distance between the source and the nearest downgradient well, as measured along a flow line is about 1,585 feet. The aquifer, which averages 50 feet in thickness, consists of gravel and sand with an average hydraulic conductivity of about 2,000 gpd/ft^2 and an average effective porosity of about .25. The difference in water level from the source to the well is 15 feet (1,500 ft-1,485ft). Therefore, the gradient is 15ft/1,585ft = .009 and the velocity is about 9.6 feet per day.

$$V = KI/7.48\ n = (2{,}000\ \text{gpd/ft}^2)(.009)/(7.48\ \text{gal/ft}^3)(.25) = 9.6\ \text{ft/day}$$

It would require about 165 days (1,585ft/9.6ft/day) for the center of mass of the contaminant to reach the well under the prevailing conditions. The rate could be slowed by reducing the gradient, that is, turning off the well or reducing its discharge. It should also be pointed out that the plume formed by the spill would arrive at the well prior to 165 days; it is only the center of mass of the plume, which is its highest concentration, that would arrive at the time calculated by this method.

An actual situation very likely would be far more complex and the travel time would be substantially less than that calculated in the above example. Nonetheless, this simple approach provides at least an estimate of what might happen.

Another approach is to use some type of graphical technique or computer model, many of which are available. It must be remembered, however, that a computer simulation can only be as valid as the equation and data on which it is based and no matter how sophisticated or expensive this simulation might be, the results can only be used as an estimate. The greatest advantage of computers is that they can provide a solution very quickly and permit one to readily change the input values in order to get a feel for the way the aquifer reacts to different stresses. Additionally, the nomograph described in Chapter 5 and computer models permit one to more accurately represent the aquifer and the manner in which it functions, particularly in regard to hydrodynamic dispersion and retardation of contaminants.

The problem described above can be solved using a nomograph or computer model, although additional data are required. These include the retardation factor (R_d) and longitudinal and transverse dispersivity (α_x, α_y). Since the contaminant in this case is chloride, a conservative chemical that is neither sorbed nor degraded, then $R_d = 1$. Dispersivity (α_x, α_y) is much more difficult to estimate. (Recall from Chapter 5 that longitudinal dispersion, D_x, equals $\alpha_x V$ and transverse dispersion, D_y, equals $\alpha_y V$). In this case, however, the distance between the contaminant source and the well is relatively short, the velocity of the ground water is high, and the water is converging in all directions toward the well bore. As a consequence, advection or simple ground-water flow is more important than dispersion and the latter either can be ignored or small numbers can be used in the equation. In the few studies available, the ratio between longitudinal and transverse

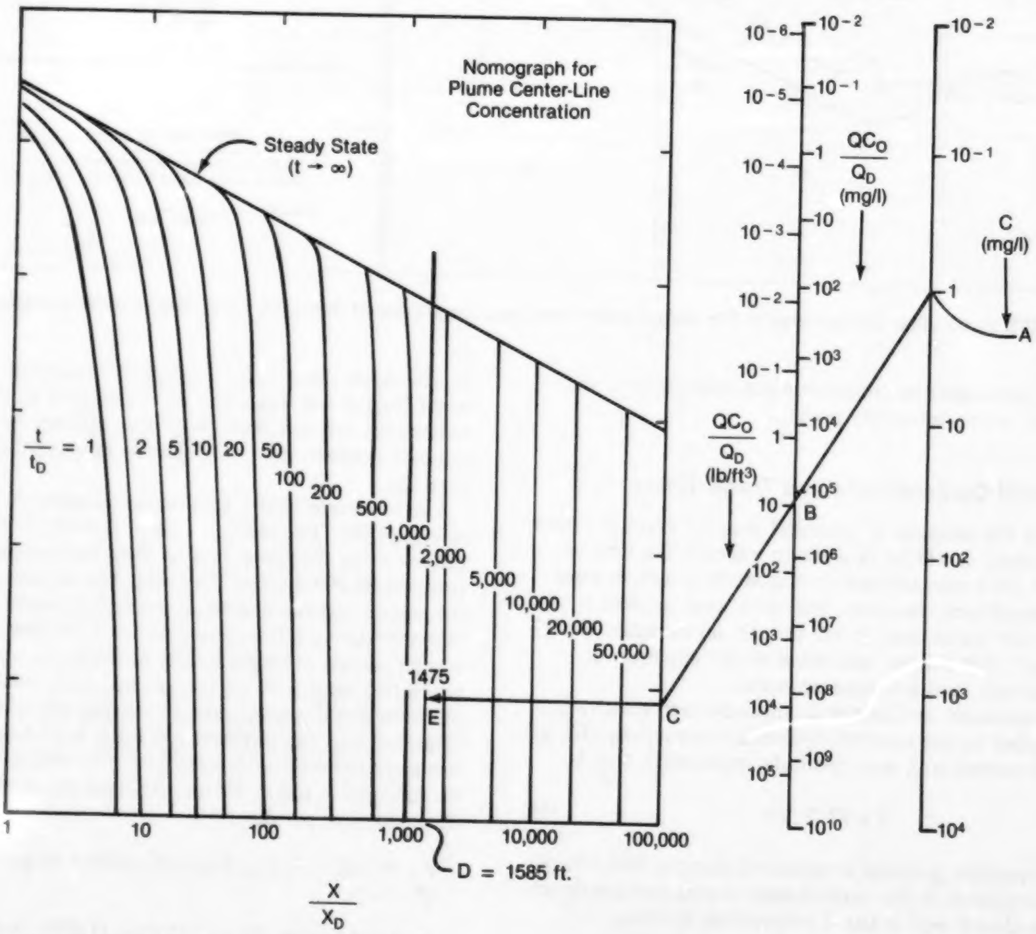

Figure 162. Nomograph Solution to the Example Problem

dispersivity ranges between 1 and 10. For this example let us assume a ratio of 5 and a longitudinal dispersivity of 1, which requires that transverse dispersivity is 0.2. Table 14 shows the data required for the nomograph solution and computer simulation.

Table 14. Summary of Data for Example

Distance from source to well (x)	1,585 feet
Aquifer thickness (m)	50 feet
Effective porosity (n)	.25
Velocity (V)	9.6 feet/day
Longitudinal dispersivity (α_x)	1 foot/day
Transverse dispersivity (α_y)	.2 feet/day
Retardation (R_d)	1
Volume flow rate (Q)	2,000 gallons/day
Source concentration (C_o)	30,000 mg/1
*Mass flow rate (QC_o)	502 lb/day

*Mass flow rate = $.134QC_o/16,019 = (.134)(2,000)(30,000)/16019$

$= 502$ lb/day

Using the nomograph equations provided in Chapter 5, calculate the following:

$$X_D = D_x/V = (\alpha_x)(V)/V = (1)(9.6)/9.6 = 1$$

$$T_D \; R_d D_x/V^2 = (1)(9.6)/(9.6)(9.6) = 9.6/92 = .1$$

$$Q_D = nm\sqrt{D_x D_y} = (.25)(50)\sqrt{(9.6)(1.9)}$$
$$= 53 \text{ ft}^3\text{/day}$$

To determine the time when the leading edge of the plume, say a concentration of 1 mg/1, reaches the well calculate:

$$X/X_D = 1{,}585/1 = 1{,}585 \text{ (Locate at D)}$$

$$QC_o/Q_D = 502 \text{ lb/day}/53 \text{ ft}^3\text{/day} = 9.5 \text{ lb/ft}^3$$
$$\text{(Locate at B)}$$

or

$$QC_o/Q_D = (267 \text{ ft}^3\text{/day})(30{,}000 \text{ mg/1})/53$$
$$= 151{,}132 \text{ mg/1 (Locate at B)}$$

Using Figure 162, locate the selected concentration (1 mg/1) at A, draw a line from A through B to C, then horizontally from C to the intersection with a vertical line from D, giving at E:

$$t/T_D = 1{,}470$$

Multiply by T_D to determine time(t), in days or

$$t = (1{,}470)(.1) = 147 \text{ days}$$

Thus, under the given conditions, the margin of the plume represented by a chloride concentration of 1 mg/1 should reach the production well in about 147 days or 18 days sooner than the center of mass calculated earlier. It is essential to remember, however, that this calculation also is only an estimate. On the other hand, it does suggest that dispersion plays an important role in chemical transport and that the contaminant will migrate faster than anticipated even with very small dispersivity values.

One of the limitations of the nomograph is that it can be used only to calcuate concentration or time directly down gradient from the source, that is, along a flow line that goes directly through the source. Another limiting factor is that the mass flow rate is constant, that is, there is no method available to stop the source from leaking, despite the fact that in the example case leakage occurred only during a 3-day period.

Several advantages over the nomograph are provided by a computer model, even though both are based on the same equation. The computer model will generate a concentration distribution map of the entire plume, not just along a single flow line. Furthermore, in addition to being fast, the model will allow the operator to insert multiple sources and vary the mass flow rate from each source.

Figures 163 and 164 show examples of computer generated maps, based on the data listed in Table 14. In Figure 161 the spill was allowed to discharge only 3 days. Notice that there is a distinct plume moving toward the well and that the concentration shown on the map in Figure 163 must be multiplied by 10. It can be calculated that, at this time (147 days), the concentration at the well is only 0.5 mg/l because the plume has not yet reached it. In Figure 164, the source was not shut off but was allowed to discharge 2,000 gpd for the entire time of the simulation (147 days), thus providing a two-dimensional view of the concentration distribution as compared to the single point nomograph solution. A

```
                              BASIC COEFFICIENTS

NO.             DESCRIPTION                                   VALUE
-------------------------------------------------------------------------
1.              VELOCITY                                    9.6 ft/day
2.              LONGITUDINAL DISPERSIVITY                      1 ft
3.              TRANSVERSE DISPERSIVITY                        .2 ft
4.              RETARDATION COEFFICIENT                        1
5.              HALF-LIFE                                   2,000 years
6.              POROSITY                                       .25
7.              AQUIFER THICKNESS                              50 ft

 SOURCE             INJECTION               START
   AT                 RATE                  TIME               LENGTH
 -------            ---------              ------              ------
  0 , 0                502                    0                  100
  0 , 0               - 502                   3                  100

          PLUME AFTER 147 DAYS                         SCALE: 10 MG/L
-------------------------------------------------------------------------------------------

  560 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  480 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  400 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  320 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  240 I   +   +   +   +   +   +   +   +   +   +   +   +   +   X   +   +   +   +   +   +   +
  160 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
   80 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
    0 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   2  19  18   2   +
  -80 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -160 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -240 I   +   +   +   +   +   +   +   +   +   +   +   +   +   X   +   +   +   +   +   +   +
 -320 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -400 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -480 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -560 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
        --+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+

          0   8   1   2   3   4   4   5   6   7   8   8   9   1   1   1   1   1   1   1   1
              0   6   4   2   0   8   6   4   2   0   8   6   0   1   2   2   3   4   5   6
                  0   0   0   0   0   0   0   0   0   0   0   4   2   0   8   6   4   2   0
                                                              0   0   0   0   0   0   0   0

SYMBOLS: X - ANSWER INACCURATE, # - ANSWER ABOVE 1000 MG/L
```

Figure 163. Computer Generated Map (Example 1)

map of this type is very misleading because it implies that high concentrations are continuing to appear between the source and the well despite the fact that the leak occurred only during a 3-day period. Notice also that the concentrations listed on the map in Figure 164 must be multiplied by a factor of 100.

```
                    BASIC COEFFICIENTS

NO.       DESCRIPTION                          VALUE
------------------------------------------------------
1.        VELOCITY                          9.6 ft/day
2.        LONGITUDINAL DISPERSIVITY               1 ft
3.        TRANSVERSE DISPERSIVITY                .2 ft
4.        RETARDATION COEFFICIENT                    1
5.        HALF-LIFE                        2,000 years
6.        POROSITY                                 .25
7.        AQUIFER THICKNESS                      50 ft

SOURCE         INJECTION        START
  AT              RATE           TIME        LENGTH
------         ---------        -----        ------
 0 , 0            502             0            100

        PLUME AFTER 147 DAYS                 SCALE: 100 MG/L
-----------------------------------------------------------------------------------------
  560 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  480 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  400 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  320 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  240 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
  160 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
   80 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
    0 I   X  47  33  27  24  21  19  18  17  16  15  14  14  13  13  12  12  10   3   +   +
  -80 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -160 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -240 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -320 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -400 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -480 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
 -560 I   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +   +
       --+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+---+

          0   8   1   2   3   4   4   5   6   7   8   8   9   1   1   1   1   1   1   1   1
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                  0   0   0   0   0   0   0   0   0   0   0   4   2   0   8   6   4   2   0
                                                              0   0   0   0   0   0   0   0

SYMBOLS: X - ANSWER INACCURATE, # - ANSWER ABOVE 1000 MG/L
```

Figure 164. Computer Generated Map (Example 2)

Situation Monitoring

Situation monitoring should cover two main categories: 1) monitoring of the existing water-supply system and plant and (2) monitoring of other local situations. The former can and should be accomplished in-house, while the latter can be carried out by interviews, the news media, and local agencies.

It is surprising that so few operators, particularly those involved with small systems, are aware of the chemical quality of their well supplies. Even if routine chemical analyses are carried out periodically, it is unlikely that samples will be scanned for the more exotic compounds, such as heavy metals or organics. This is understandable in view of the cost. On the other hand, without background data it is commonly difficult, if not impossible, to detect many contaminants or locate a source, especially if proof is required in a legal action. The costs of chemical analyses must be accepted by the operator as a part of the business practice.

Another part of the survey that should be conducted by the water utility personnel includes an examination of their facilities addressing such items as: (1) possibility of back siphonage, (2) cross-connection, (3) distribution system deficiencies, and (4) poor well construction or location. Are there, for example, potential sources of contamination, such as fuel tanks or sewer lines, adjacent to the well or well house? In the fall of 1971 at a trailer court in Anchorage, Alaska more than 80 individuals became ill due to consumption of sewage contaminated well water obtained from a semi-public supply. The system consisted of 2 wells, about 242 feet deep, enclosed in a block well house. A soft plug formed in a borough sewer causing raw sewage to back-up, eventually to flow from a drain in the floor of the well house and, when reaching a foot in depth, to flow directly into and down the well casing. Subsequently, sewage contaminated water was pumped into the water supply of the trailer park.

Another part of situation monitoring involves the collection and evaluation of information in the communty or area within the influence of the cone of depression of the well field. For example, have there been any fires that might have resulted in the release of chemicals that could reach the aquifer? Have there been any spills from truck or train wrecks? Might new construction produce a hazard? Are plans being developed for the placement of hazardous waste storage or disposal sites? In other words, the purpose of surveys of this nature, which must be continuous, is to keep in touch with the community.

Alternate Sources of Supply

A common solution to a water-quantity problem is to deepen a well and to a contamination event is to offset and drill another well. Unfortunately, such potential solutions, although simple, are rarely viable. It may not be possible to deepen a well and merely offsetting a contaminated well will only solve the problem for a few hours or days. It appears that human nature is such that we tend to procrastinate, hoping that life will continue uninterrupted. The far thinking individual, however, will consider alternatives, formulate cost estimates, and develop plans, both for design and obtaining the necessary funds for construction, for other sources of supply.

Is there a source of surface water available nearby that will meet water quality standards after treatment? If so, is a site available and what are the potential costs of constructing intake structures and conveyance facilities and of treating the water, and how much time would it require to actually provide the water? Is the supply dependable, or contaminated, or can water rights be obtained?

Sometimes it might be possible to use nontraditional concepts to develop a surface supply. One method might be to use collection gallaries, particularly if the available streams are small. In this case a ditch could be cut across the stream into which a gravel bed is placed. A well screen attached to a suction line can be placed

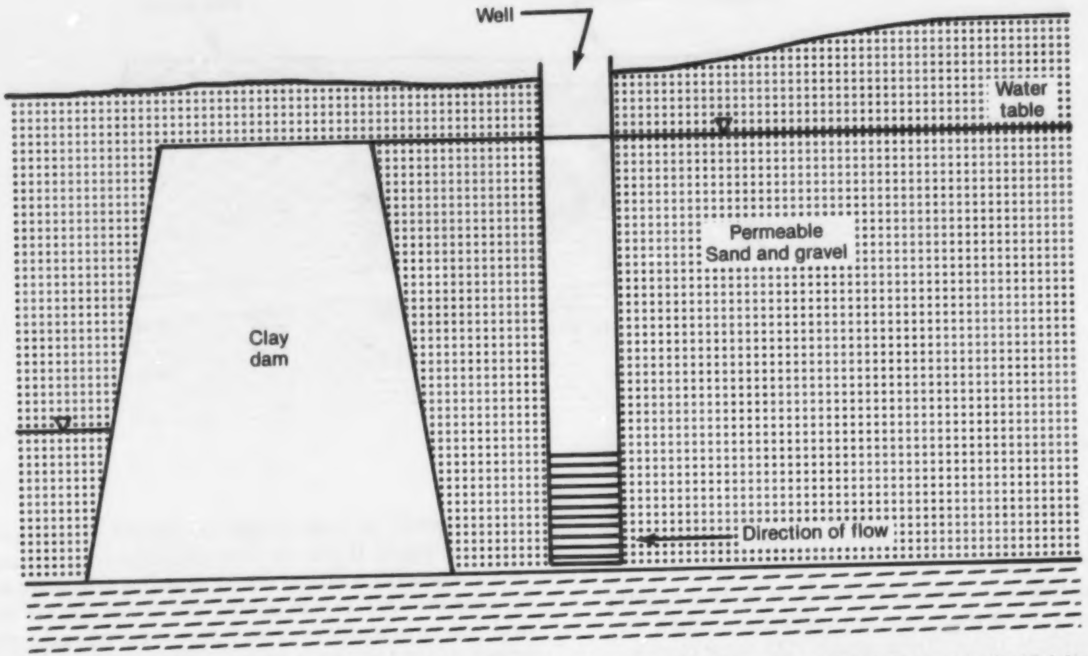

Figure 165. A Subsurface Dam of Clay Impeded the Flow of Ground Water in a Pond and Gravel-Filled Channel in the Vicinity of Glenburn, North Dakota.

on the bed and the remainder of the ditch filled with gravel. This is virtually a horizontal well whose supply depends on infiltration of surface water through the gravel pack. Although the filter pack might well reduce turbidity, it would have little or no effect on many chemical contaminants. Nonetheless, this technique offers a simple and relatively inexpensive altenative.

Another method is the subsurface dam. The village of Glenburn in north-central North Dakota had a difficult time supplying sufficient water for their needs. They overcame this deficiency with an unusual and inexpensive artificial recharge technique. Most of the surficial rocks in the Glenburn area consist of clay, but nearby there is a 30 foot wide channel, usually dry, that contains 7 to 8 feet of coarse gravel and sand. Upstream the deposit widens and there is an abandoned gravel pit.

During the spring runoff a considerable amount of water infiltrates the gravel and the water table rises dramatically. Because the deposits are very permeable, the ground water flows down gradient quickly, however, and the water table soon declines as the aquifer is drained. The gravel channel has a considerable capacity for storage but no natural controls to prohibit rapid drainage.

As Figure 165 illustrates, this problem was solved by excavating a ditch, 4 or 5 wide, across the channel and entirely through the gravel deposit. The excavation was backfilled with readily available clay forming a subsurface dam. A perforated culvert, serving as a well, was installed on the upstream side. A diversion ditch was excavated from the intermittent stream to the abandoned gravel pit, which served as a recharge basin. During periods of runoff, some of the surface water flows into the gravel pit, where it infiltrates, and part of the remainder infiltrates along the stream bottom. Thus, during wet periods a considerable amount of water is collected in the underground storage reservoir. The subsurface dam impedes the flow of the ground water down the gravel-filled channel and the water table remains at a high level, permitting increased water usage. It should also be mentioned that this was a community project accomplished by volunteer labor and equipment. The total cost was minimal.

Water supply problems in arid regions are particularly vexing because of scanty rainfall and the high rate of evaporation. In some situations, it may be possible to augment supplies by constructing artificial aquifers. Artificial aquifers, by necessity, can store only moderate quantities of water, but they are labor intensive and, therefore, can be built at a modest equipment cost.

The design and construction of an artificial aquifer at the Santa Clara Indian Reservation, New Mexico, was described by Helweg and Smith.[74] As Figure 166 shows, a small gulley, several yards wide, was cleared of vegetation, deepened, and sloped. Spoil material was used to construct an earthen dam across the gulley. A trench was cut adjacent to and parallel with the dam into which was installed a slotted plastic pipe. The slotted pipe was connected, at a right angle, to a second pipe, extending through the dam in the low point of the gulley. The second or discharge pipe was laid on a slight downslope and installed prior to dam construction.

Once the gulley was shaped, the pipes installed, and the dam built, plastic lining was placed on the floor of

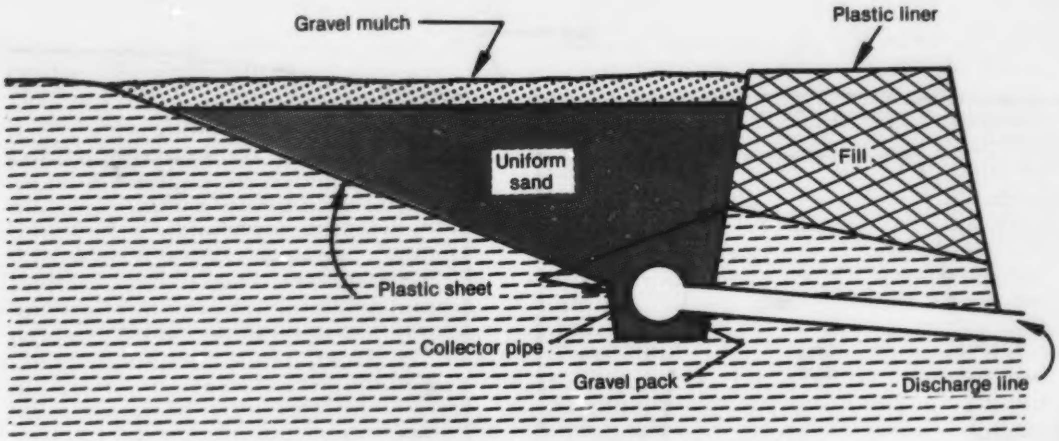

Figure 166. Schematic of an Artificial Aquifer

the structure. The gulley was then backfilled with uniform sand (gravel could be used) and topped off with gravel mulch.

During the rainy season or periods of surface runoff, water flows down the gulley and infiltrates through the gravel mulch to the artificial aquifer. (In some cases it might be necessary to construct a spillway to avoid excessive erosion of the dam.) Water is removed from the reservoir via the discharge pipe, the rate being controlled by a valve.

Another management alternative is to consider developing another aquifer or a different part of a contaminated aquifer. In the latter case, care would be required in the well field design to insure that the new system would not be contaminated due to changing hydraulic gradients.

Generally, new well fields require considerable time and financing to achieve a proper design and adequate construction. The first question to be addressed should be, "Is there an aquifer available that will supply the required needs and what are its characteristics?" If one is available, what is the quality of the water it contains and what is an estimate of the treatment requirements and costs? How many wells will be required and how much will they cost?

Probably one of the most unusual and farsighted ground-water systems was designed in Kalamazoo, Michigan, which in the 1930's was suffering from a severe water shortage. One individual who worked for the city, Mr. Al Sabo, took a leave of absence to work for a drilling firm in order to learn the trade. After several months he returned to his previous position and convinced city officials that a drilling rig should be purchased. A crew then began to drill test holes throughout a wide area, paying particular attention to swampy regions. In this glaciated terrain, streams are commonly small but they flow throughout the year, that is, the flow is sustained by ground-water runoff. Furthermore, swampy areas characterize places where the water table is at or near the land surface and in the Kalamazoo drilling program it was found that these swampy areas were underlain by considerable thicknesses of saturated sand and gravel. If the test hole encountered a substantial thickness of sand and gravel, a well screen and casing were installed and an aquifer test conducted. If the yield was satisfactory the pump was removed and a plate welded to the top of the casing until the well was needed. The swampy area was then purchased by the city and the sand and gravel sold, which more than paid for the property and wells. After the sand and gravel were mined, the excavation served as an artificial recharge basin. In a few years, the city of Kalamazoo had more than enough water to supply all of their needs, plus all the water required by several surrounding communities.

Periodically various regions suffer from prolonged droughts, streamflow decreases or may even cease, and water rationing becomes the rule. It is interesting to note that in many of these areas it is only the surface-water supply that is decreasing, while billions of gallons of ground water remain untapped in naturally occurring underground reservoirs that remain hidden from view. This, of course, was the case in Kalamazoo in the early 1930's.

Legal Controls on Waste Disposal

A variety of laws, regulations, and rules exist to control waste disposal. In addition to the often quoted federal laws, there are laws established by state legislatures and regulations formulated by state agencies. Local zoning ordinances may play an important role for the water-plant operator. These need to be researched, understood, and modified as the need arises.

Development of an Aquifer Protection Plan

The basis of an aquifer protection plan is at least a general knowledge of the aquifer system and the manner in which it functions. Where are the inherent weaknesses of the physical system, where are the strong points, and where is a contamination problem most likely to occur? These questions can best be answered or at least evaluated by means of a series of maps.

The maps shown in Figures 167 to 170 are modifications of illustrations published some years ago in a report by the U.S. Geological Survey.[75] Many similar maps, reports, and books have been published by federal, state, and local agencies. In cases where none are available, the utility operator may be required to prepare his own, hopefully with the aid of an accomplished hydrogeologist. The data base, of course, consists largely of well logs and water-level measurements.

The map in Figure 167 shows a large cone of depression around a well field and the arrows indicate the general direction of ground-water flow. Notice that the contours indicate that water is flowing into the pumping center from all directions. Consequently, a large area should be protected or at least monitored because contaminants reaching the aquifer in any part of the area of pumping influence could eventually reach a well.

The map in Figure 168 plots potential sources of contamination. Information of this type can be obtained from the waste surveys discussed earlier. All fixed sources are plotted, as well as the location of major transportation routes, such as railroads and highways.

Figure 169 outlines the degree of natural protection afforded the aquifer. The map, based on well logs, indicates that the eastern part of the valley-fill aquifer is protected by a considerable thickness of clay through which contaminants are not likely to flow. To the west, however, the valley fill consists of sand and gravel that extends from the land surface to bedrock, a distance of more than 100 feet. Any contaminants entering the ground here could quickly reach the aquifer. In the central part of the aquifer the major water-bearing zone has some overlying protection in the form of alternating layers of clay and sand. Although a contaminant eventually could reach the aquifer in this area, it would require a substantial amount of time and probably the contaminant would be degraded or sorbed to some extent and certainly diluted as it migrated through several tens of feet of clay and sand.

Figure 170 shows an aquifer sensivity map. Based largely on well logs and the previous map, it rather

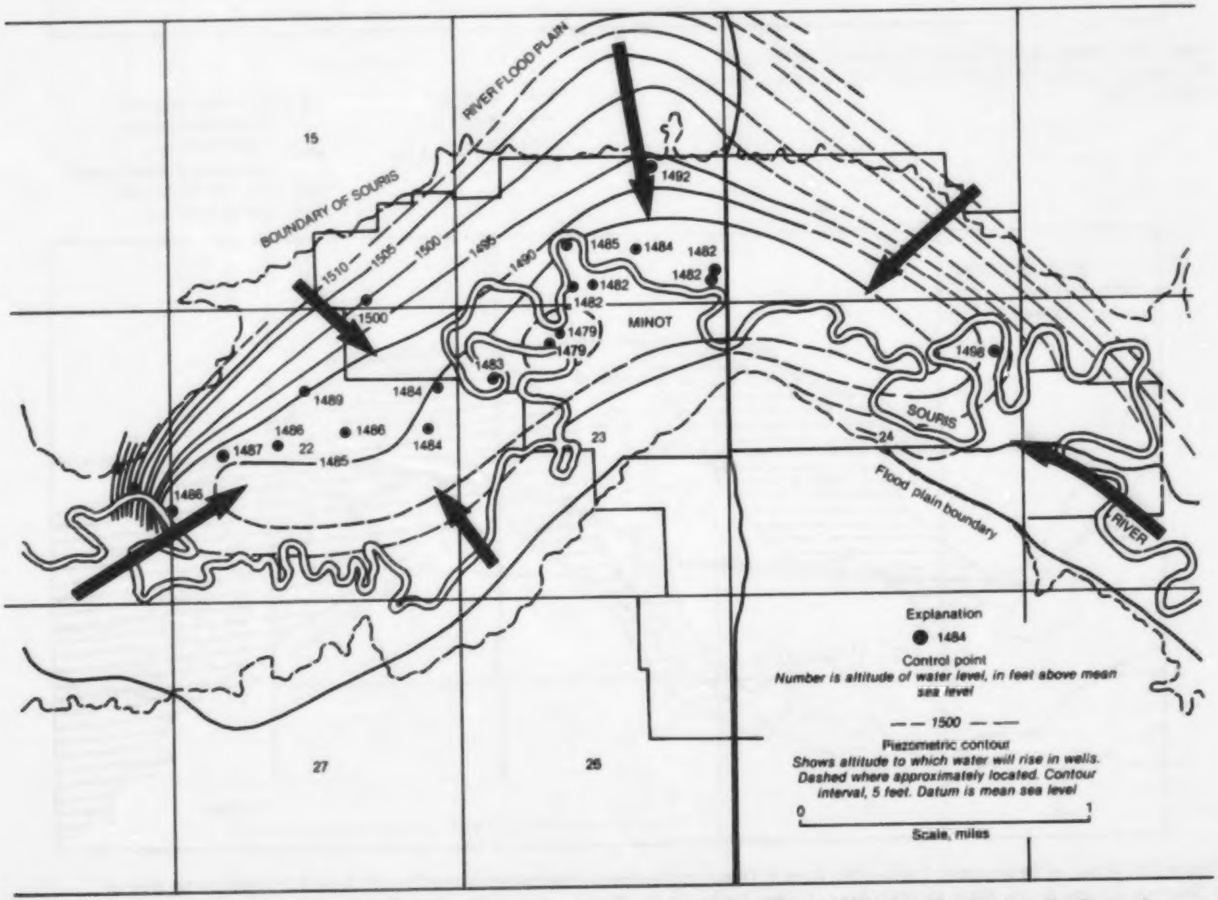

Figure 167. Water-Level Contours Indicate that the Cone of Influence Surrounding the Well Field Extends Outward for Miles in the Valley and that Water is Flowing Into the Cone of Depression from all Directions.

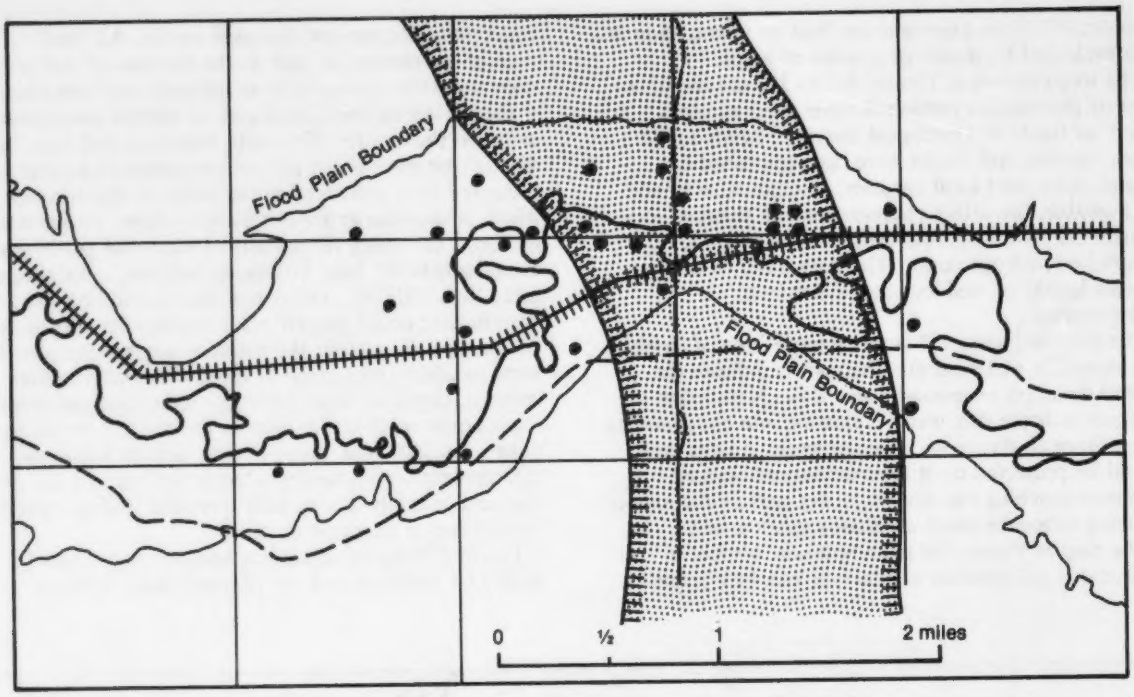
Flood Plain Boundary
Flood Plain Boundary
0
½
1
2 miles
Explanation
Railroad
Highway
Buried Valley
Potential sources of contamination
Potential line source of contamination
Source of natural contamination

Figure 168. Potential Sources of Contamination Include Highways, Airports, Railroads, and Streams, as well as Commercial Sites.

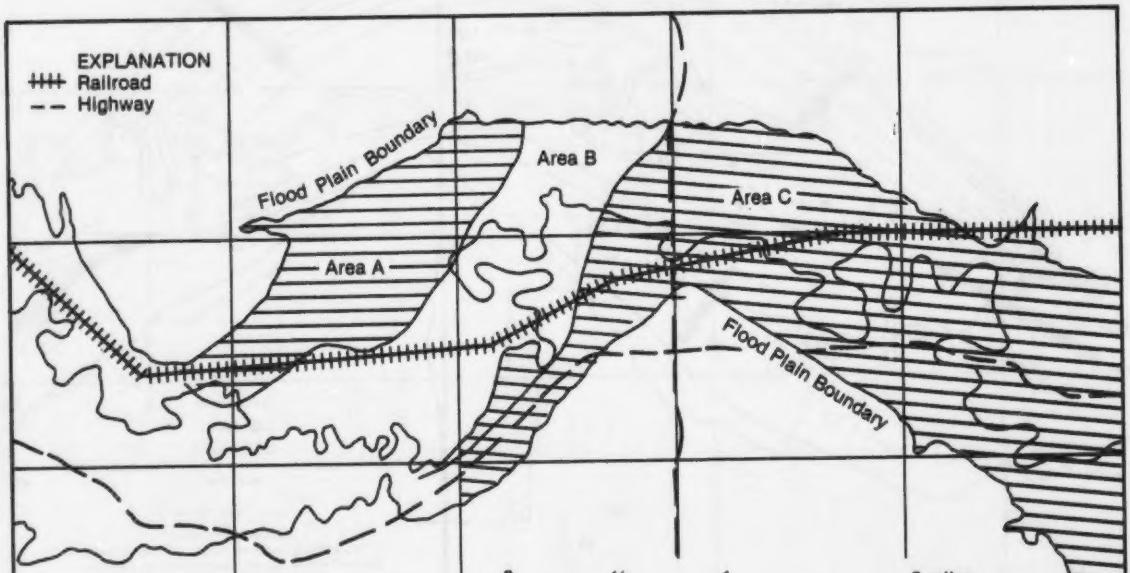

Figure 169. Areas of Recharge to the Aquifer. Area A, Direct Infiltration of Precipitation and Runoff. Area B, Infiltration by Way of a Shallow Aquifer to the Main Aquifer. Area C, Area Underlain by Thick Layers of Clay Resulting in Only Small Quantities of Recharge.

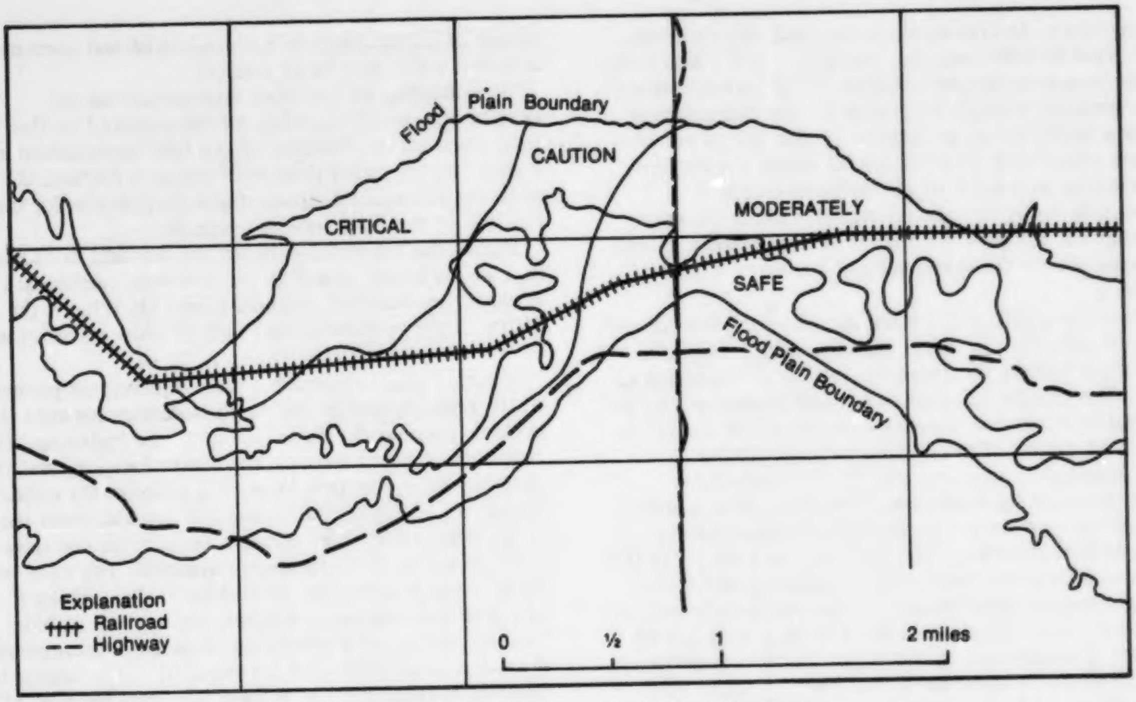

Figure 170. An Aquifer Sensitivity Map Indicates the Range of Natural Protection of the System.

clearly indicates those areas of most and least concern. The eastern part of the aquifer is moderately safe, caution should be exercised in the central part, and the western part should be carefully protected and monitored. The latter critical area should be brought to the attention of city officials and an attempt made to protect it by local zoning ordinances.

Emergency Response Plans

This section addresses what action a water utility may be required to take as a result of contamination of the ground-water source used to provide public drinking water. It must be emphasized, however, that the prevention of ground-water contamination, through proper management of the resource, is the least expensive approach. The old adage of "an ounce of prevention is worth a pound of cure" is certainly applicable to this vital resource. During this seminar series, we have not seen many examples of planning for prevention of ground-water contamination or, for that matter, of a planned course of action for loss of the primary water supply.

When ground-water contamination is confirmed, there are two immediate problems: (1) notifying the consumers and (2) providing a safe drinking-water supply. For most utilities, the task of notifying the public that their drinking water has been contaminated will be the most difficult activity in responding to the contamination. In transmitting its message to the public, the utility must answer the technical questions of how it will respond to the situation as well as the more personal question of the consumer of, "What does this problem mean to me?"

There is no sure-fire plan which can be followed that will prevent an adverse response from the public to this message. There are a few points that have been borne out through the case histories we have heard in this series. The most important of these points are: (1) tell the public the truth as you know it, (2) tell them as soon as the presence of contamination has been confirmed, and (3) continue to keep them abreast of the situation and any remedial actions you are taking.

Only through these three steps do the people dealing with the situation have any hope of maintaining credibility with the public. Any attempt to hide or gloss over the situation is an invitation to an adverse public response and potential litigation.

The next problem—continuing to provide a safe potable drinking-water supply—presents the utility several alternatives based on the particulars of the situation. The possibility of a timely and logical response is greatly enhanced by prior planning for dealing with loss of the ground-water source. This response plan should be developed in sufficient detail to be implemented with only fine tuning to address the specifics of the particular incident. Agreements between the various parties, which outline the responsibilities of each party, should be laid out in advance and not postponed until the emergency is at hand. Response to the ground-water contamination may range through a broad spectrum from issuing a boil-water order to providing an alternative drinking-water supply. In considering a response, the entire picture must be addressed. For example, "Will a boil water order be a good interim solution or will exposure to the contaminants through inhalation or skin contact be as great a health threat as ingestion?" The use of an alternative source may create problems within the distribution system as a result of the different chemical characteristics of the new source. If water is provided through the use of water trailers or bulk haulers, efforts to maintain the potable quality of the water must be taken.

Once the public notification and the continued provision of safe drinking water has taken place, the utility may then address the long-term problem of remedial action. The specific response that a utility takes will be impacted by a number of factors among which are: (1) the type and characteristics of contaminant; (2) the hydrogeologic conditions; and (3) the political, economic, and legal concerns. Therefore, how a utility should respond cannot be detailed in a presentation such as this. However, there is some commonality in the steps which can be taken when considering alternatives. These steps do not necessarily occur sequentially and, in fact, may occur simultaneously. The steps may not be entirely accomplished by the utility, but it is important for the utility to understand the approach taken by its consultants or the authorities addressing this problem. Activities can be placed in 4 groups: (1) gathering and evaluating available information, (2) developing the field investigation to provide unavailable information, (3) developing and selecting remedial alternative(s), and (4) dealing with state and federal regulatory authorities and the public.

The first two areas, gathering and evaluating available information and developing the field investigation, are closely linked. The full development of the field investigation should only occur after the available information has been gathered and evaluated. This will allow maximum use of the finite funds available for gathering information.

In gathering and evaluating available information, we are looking to estimate the extent, nature, and direction of movement of the contamination. This estimate can be based on readily available information from such agencies as the U.S. Geological Survey and state geological organizations. Topographic and hydrologic maps will provide insight into geologic units and influence of landforms and surface waters on ground water. The characteristics of the contaminant(s) and a knowledge of industrial/commercial activities in the area may help in locating the source of contamination. Development of a map overlay depicting potential source(s) such as gasoline stations, industrial sites, highways, and railways will assist in this search. We frequently limit our concern to industrial/commercial activities; however, we should not overlook the impact of domestic activities and disposal practices. Cases involving contamination of private wells with materials such as chlordane have created widespread contamination of an aquifer. State regulatory and health agencies can be a source of information on both industrial and domestic activities which may be of concern.

By evaluating the available information we can develop the questions which will be answered by the field investigation. Developing the field investigation in a logical manner rather than undertaking a haphazard program of installing monitoring wells provides for the best use of the finite monies available.

During the information gathering and field investigation we are trying to answer the following questions: (1) What is the source of contamination? (2) What is the nature of the contamination? and (3) How extensive is the contamination?

We are trying to develop a three-dimensional picture of the area. By understanding the interplay between the geologic framework, the topography, the hydrology, the contaminant characteristics, the source location, and the drinking water well field location a program for remedial action can be developed. A word of caution about this information cycle; there comes a point when you have to act based on the information available. The cycle of gathering of information, evaluating it, determining if sufficient information is available upon which to base remedial action, and continuing to develop information if there is insufficient data for remedial action cannot be repeated indefinitely. There have been case histories of the cost of data gathering, in particular the field investigation, exceeding the cost of the remedial action. The cost of gathering additional information and any resultant delay in implementing remedial action must be balanced with any benefit that might be gained by having that information.

Based on the knowledge of the contaminated area determined in the information gathering phase, a remedial response can be developed. What is an appropriate response? The remedial program will be unique for each site because of the influence of the type of contaminant, the geology, the economics, the politics and the legal aspects of that specific site. The final selection will be based on development of alternatives, analysis of those alternatives, and selection of the best overall alternative for the situation. In the development of alternatives, a number of alternatives, which upon initial evaluation appear applicable, are selected. After preliminary assessment of each of these alternatives, the list of alternatives is narrowed.

The alternatives remaining after this screening should be developed in more detail. The construction specifications should be stated and detailed cost estimates should be made. Additional data gathering which includes laboratory pilot scale studies may be required. Based on this detailed information the best final selection can be made. It is important to understand that the best overall alternative may not be the "best" technical solution because of the impact of political, economic, and legal constraints.

Some of the alternatives which may result are:

(1) No response — the ground-water contamination is not projected to be a threat. Additional monitoring of the situation may be all that is required.

(2) Inground Treatment/Containment
(3) Onsite Treatment
(4) Alternative Sources

The areas of inground treatment/containment and onsite treatment will be developed later in the publication. However, a brief comment on the use of alternative water sources is in order here. Although an alternative water source is generally an immediate solution to the contamination of a well field, it is frequently the final alternative selected. The reason for this can be: (1) economic constraints and (2) technological constraints.

The response to Federal, state, and local agencies may be an immediate concern and will last throughout the rest of the problem. It should be realized that in some states as many as eight agencies have responsibility for some aspects of ground water. When you add the groups within the Federal EPA responsible for various regulations such as the Resource Conservation and Recovery Act, the Toxic Substances Control Act, and the Safe Drinking Water Act, you can begin to understand the number of people that may be interested in your ground water contamination problem. You should also realize that your objective of providing a safe, potable drinking-water supply may be only one concern among many in responding to the problem. Concerns for cleaning up the site, pursuing enforcement action, securing the spill site, cleaning up surface waters and soil may be among the other objectives. In dealing with these organizations, a number of items must be accomplished which include: (1) establishing who is responsible for what, (2) developing a relationship of trust among the players, and (3) for the water utility, assuring that provision of a safe drinking water remains one of the top objectives.

In conclusion, it should be pointed out again that the prevention of ground-water contamination is a better program than responding to the immediate and sometimes gut-wrenching requirements of a contaminated well field.

References

[74] Helweg, O.J. and G. Smith. 1978. "Appropriate Technology for Artificial Aquifers". *Ground Water*, Vol. 16, No. 3, pp 144-148.

[75] Pettyjohn, W.A. 1967. "Geohydrology of the Souris River Valley in the Vicinity of Minot, North Dakota". U.S. Geological Survey Water-Supply Paper 1844.

Chapter 7

Controlling Volatile Organic Compounds in Ground Water Used for Drinking

Introduction

While numerous contaminants may be present in ground water, e.g., inorganic, microbiological, organic, radiological, this paper will discuss only organic contaminants and give an *overview* of the options available to a water utility to lower the concentrations of these contaminants to acceptably low levels for drinking. Further, this paper will discuss only the volatile organic compounds (VOCs) listed in Table 15. Some of these VOCs have been found throughout the United States in ground waters.[76] In 1982, the first step of the regulatory process was undertaken to limit exposure of these compounds in drinking water.[77] This paper relates to drinking water sources which typically contain less than 1,000 micrograms per liter (μg/l) of total VOCs as distinguished from hazardous waste disposal sites where VOC concentrations may exceed several milligrams per liter (mg/l).

Table 15. Volatile Organic Compounds Found in Ground Water Used for Drinking[76]

trichloroethylene*
tetrachloroethylene*
carbon tetrachloride†
1,1,1-trichloroethane†
vinyl chloride*
cis-1,2-dichloroethylene*
trans-1,2-dichloroethylene*
benzene††
chlorobenzene††
dichlorobenzene(s)††
1,1-dichloroethylene*
1,2-dichloroethane†
methylene chloride†
trichlorobenzene(s)††

---

*alkene (unsaturated compound)
†alkane (saturated compound)
††aromatic

Mention of commercial products in this chapter does not constitute acceptance or endorsement by U.S. EPA.

For the sake of brevity, acronyms are sometimes substituted for VOC names. For example, TCE is a recognized name for trichloroethylene, but, unfortunately,TCE has sometimes been used in newspaper articles and some engineering reports to refer to tetrachloroethylene or 1,1,1-trichloroethane. Similarly, DCE is confusing because it has been used to refer to any of the three isomers of dichloroethylene. This chapter will use the entire VOC name to avoid any confusion.

When a water utility is confronted with VOC contamination it should systematically approach a resolution of the problem. This involves characterizing the problem, analyzing the data describing the problem, and considering a site-specific solution to the problem.

Characterization of the Problem

When VOC contamination is discovered in a distribution system sample, the first step the water utility should take is to determine which, and particularly how many, contaminants were identified in the water sample. The reason is that a single compound, such as trichloroethylene or tetrachloroethylene, found in a distribution system sample can be a materials problem. For example, vinyl-toluene-lined asbestos cement pipe has been used in certain parts of the country; tetrachloroethylene leached out of these liners and into the drinking water.[78] Trichloroethylene has been detected in joint solvent used for reservoir liners.[79] Certain organics from contaminated soil can penetrate plastic pipe.[80] On the other hand, a distribution system water sample that contains several VOCs is generally a good indication that the water source is contaminated. The Ground Water Supply Survey[77] found that an affected ground water typically contains several VOCs, with trichloroethylene and tetrachloroethylene being the most common. Comparison of the VOCs found in a contaminated well and those used or previously used in the community may lead to clean up at the source. Identification of a materials problem may lead to material changes, rather than to implementation of water treatment processes.

Table 16. Hypothetical Well Water Analysis

Total Organic Carbon (TOC)	0.75 μg/l
Trichloroethylene	500 μg/l
Tetrachloroethylene	200 μg/l
cis-1,2-Dichloroethylene	100 μg/l
1,2-Dichloroethane	10 μg/l
Total Organic Halogen (TOX)	700 μg/l as Cl^-
Iron	1.0 mg/l

When a problem is discovered in distribution system samples, all the wells in the system should be sampled and, if possible, a hydrologic investigation should be undertaken to define movement of VOCs within the aquifer and their relationship to each well. A contam-

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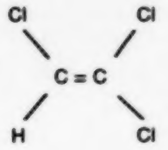


Trichloroethylene

MW = 131.5

Trichloroethylene = 500 µg/L

Carbon: $\frac{2 \times 12}{131.5} \times 500 = 91$ µg/L

Hydrogen: $\frac{1 \times 1}{131.5} \times 500 = 4$ µg/L

Chlorine: $\frac{3 \times 35.5}{131.5} \times 500 = 405$ µg/L

TOX = 405 µg/L

TOC = 91 µg/L = 0.091 mg/L

Figure 171. Example of Calculating Organic Carbon and Organic Halogen Contributions for Trichloroethylene

inated well should be sampled not only for VOCs, but also for other parameters. Table 16 is a hypothetical well water analysis showing a typical blend of VOCs and other parameters pertinent to the design of treatment processes.

Analysis of the Data

It is important to analyze the water-quality data describing the problem. The total organic carbon (TOC) concentration is a very good overall indication of organic quality. Well water typically contains 1 milligram per liter (mg/l) or less of background organic carbon, but some highly colored ground waters have a TOC concentration above 10 mg/l. Waters with high TOC may decrease the capacity of granular activated carbon (GAC) to adsorb VOCs. Note that the units for TOC and iron are milligrams per liter, whereas the volatile organic compounds are expressed in micrograms per liter (µg/l). Iron can precipitate on GAC and on the packing material in aerators and foul their operation.

It is useful when interpreting such data to determine how much of the TOC and total organic halogen (TOX) can be accounted for with specific compounds. Figure 171 is an example of this calculation using trichloroethylene. Table 17 is a summary of the accountable carbon and halogen. Note, only about 20 percent of the background organic carbon can be accounted for in this water sample. On the other hand, the accountable organic chlorine is slightly less than the TOX concentration. This might signal further investigation for halogenated compounds because the TOX concentration would be expected to be approximately 25 percent *less* than the sum of the chlorine concentration of the individual compounds[81] if the only organic halide contribution were the VOCs.

When analyzing the data, high concentrations of trihalomethanes (THMs) might be viewed with suspicion. Chlorinated well waters are typically low in THMs and THMs seldom occur in well water itself. It is not uncommon for laboratories to misidentify VOCs as THMs because of similar gas chromatographic retention times.

Table 17. Summary of Accountable Organic Carbon and Organic Halogen

VOC	Concentration, µg/l	TOC mg/l	TOX µg/l
Trichloroethylene	500	0.091	405
Tetrachloroethylene	200	0.029	171
cis-1,2-Dichloroethylene	100	0.025	73
1,2-Dichloroethane	10	0.002	7
	Calculated Total	0.147	656
	Measured	0.75	700

Solutions to the Problem

Of the two solutions offered for contending with VOC contaminated ground water, non-treatment techniques will be discussed first.

Non-Treatment Alternatives

Take the well out of *potable* service. The word potable is italicized because wells have been pumped to waste to prevent contamination from spreading to other wells in the aquifer. Shutting down a well may cause the contamination to move to uncontaminated wells.

Locate and stop the source of the VOCs. This is, of course, desirable, but pinpointing the source is generally difficult and often impractical. Complicating factors are biological or abiotic transformations. A solvent such as trichloroethylene may have been the original organic chemical that was spilled, discarded, or disposed of, but because of certain biological conditions within the soil or because of chemical hydrolysis reactions[82], transformation products that ultimately reach the well may be vinyl chloride or cis-1,2-dichloroethylene.[83-85]

Blend well water. All water purveyors are familiar with this alternative to improve inorganic quality by moderating hardness or diluting the total dissolved solids from one or more wells. VOC and non-VOC contaminated wells have been successfully blended both with and without hydraulic changes in the distribution systems.

Alternate Sources. Generally, the first course of action taken by a utility when VOC contamination is discovered is to drill a new well. A review of the pros and cons of drilling a new well is given by Gaston.[86] In cases cited by Gaston, a new well could mean investing several hundred thousand dollars with no guarantee that the new well would remain uncontaminated after development. Again, hydrogeologic investigation may reveal whether drilling is a viable alternative.

Interconnection. A connection to a neighboring distribution system or source exists at many utilities. If not, making such arrangements for emergency preparedness is prudent. This alternative is similar to blending.

Bottled water. This alternative can provide some immediate action while a permanent solution is being investigated. However, this alternative suffers from the same shortcomings as some point-of-use devices in that VOC inhalation exposure from showering is neglected.

Treatment Alternatives

Conventional Water Treatment. In general, conventional water treatment (i.e., coagulation, flocculation, sedimentation, and filtration) is not effective for removal of VOCs. Following spills, VOCs were found to pass through surface water treatment plants with little removal.[87, 88] Any losses were likely the result of evaporation from open basins.

Oxidation and Ultraviolet Irradiation. The U.S. EPA's Drinking Water Research Division (EPA-DWRD) found no reduction in VOC concentration following treatment with chlorine, chlorine dioxide, or hydrogen peroxide.[89]

Using permanganate ion or ferrate ion, no control of the alkanes or the aromatics listed in Table 15 was observed. The alkenes listed in Table 15, however, were oxidized by these ions. EPA-DWRD found oxidation decreased as halo-substitution of the alkene increased. Tetrachloroethylene, for example, was unaffected by permanganate, but trans-1,2-dichloroethylene and vinyl chloride were rapidly oxidized by 2 mg/l permanganate. No attempts were made to identify end products which may also be halo-substituted.[89] The reaction between alkenes and permanganate is well known.[90]

EPA-DWRD found ozone to be ineffective for oxidation of alkanes, but effective for control of aromatics and alkenes.[89] The end products of these oxidation reactions also remain unknown.

Ultraviolet irradiation was found by EPA-DWRD to reduce VOC concentrations. Under similar conditions, better removals of alkenes and aromatics than alkanes were observed. When ultraviolet irradiation was combined with hydrogen perioxide oxidation, little or no improvement was observed when compared to ultraviolet treatment alone.[89] Several commercial processes are available combining ultraviolet irradiation and ozonation. The mechanisms and end products of these processes are not well defined.

In summary, certain VOCs are not influenced by oxidants, yet others are to a degree. VOCs are probably not oxidized to carbon dioxide, halide, and water; rather, end products result that, as yet, are uncharacterized. These processes are likely more costly than other alternative, e.g., aeration or adsorption, and may not be suited to a typical blend of VOCs which includes alkanes.

Reverse Osmosis. EPA-DWRD found a spiral-wound cellulose-acetate membrane to be ineffective for VOC removal. A hollow fiber, polyamide membrane performed somewhat better; it rejected about 25 percent of certain VOCs.[91] VOC rejection tests, however, on thin-film composite membranes demonstrate better control.[89]

It was felt that if VOCs were discovered at sites where reverse osmosis was already being used to control inorganic compounds, the best alternative may be simply to change to a thin-film composite membrane. Unfortunately, EPA-DWRD discovered that a short-term experimental protocol can be deceptive. Note in Table 18 that several VOCs diffused through a thin-film composite membrane after a few hours. Carbon tetrachloride and 1,1,1-trichloroethane were well rejected. Subsequent studies of other thin-film composite membranes showed similar patterns of time-dependent performance and better control of some VOCs than others.[89]

In summary, this process may be useful for treating certain VOCs. Prior testing should evaluate the membrane over several weeks time. This process may be more costly than other alternatives. Finally, consideration must be given to proper handling and disposal of the concentrated reject stream.

Table 18. Removal of Volatile Organic Compounds* by a Thin-Film Composite Reverse Osmosis Membrane[91]

Organic Compound	Average Influent conc., μg/l	Percent Removal				
		1 hr.	3 hrs.	6 hrs.	9 hrs.	21 hrs.
Benzene	39	52	20	18	18	19
Carbon tetrachloride	109	98	98	98	98	98
Chlorobenzene	23	98	96	85	72	50
1,2-Dibromomethane (EDB)	8	88	34	39	15	15
m-Dichlorobenzene	26	95	91	88	85	66
o-Dichlorobenzene	36	93	88	85	83	70
1,2-Dichloroethane	50	49	34	29	29	37
cis-1,2-Dichloroethylene	65	47	6	24	12	20
Tetrachloroethylene (PCE)	95	99	98	97	74	68
1,1,1-Trichloroethane	90	98	98	98	98	98
Trichloroethylene (TCE)	97	89	66	35	25	30

*Spiked in distilled water.

Aeration. One inquiry showed approximately three dozen water utilities in the United States were using aeration to alleviate problems with VOCs in their ground water.[92] As noted, the treatment alternatives presented above are more costly than certain types of aeration and adsorption. A more extensive discussion on these technologies is therefore warranted. Much of the following aeration discussion is taken from reference 92.

Aeration can be broadly categorized in two ways: (1) putting air through water and (2) putting water through air. In the air-through-water systems, mass transfer takes place at the bubble surface and the practical upper limit for the air-to-water ratio is probably around 20 to 1 (volume-to-volume). The water-through-air systems create water droplets or a thin layer of water to facilitate mass transfer. Although air-to-water (A/W) ratios as high as 3000 to 1 (vol-to-vol) have been reported[93], the A/W ratio is generally less than 100 to 1.

Air-Through-Water Aeration Systems

Diffused-Air Aeration. As Figure 172a illustrates, in diffused-air aeration compressed air is injected into water through perforated pipes or porous plates. These are sometimes called injection or bubble aerators and are more commonly used for transferring oxygen into waste water. Figure 173 shows the results of a one-year pilot-scale study at a contaminated ground-water site where the aerator removed approximately 95 percent of the total volatile organic compounds. Figure 173 shows the results of a one-year pilot-scale study at a contaminated groundwater site where the aerator removed approximately 95 percent of the total volatile organic compounds. Figure 173 is typical of any type of aeration device in that VOCs are generally not removed to "below detection" levels. One advantage of diffused-air aeration is that a system can be put into operation very quickly using existing facilities. Reservoirs, caisson wells, and even well casings and well bores have served as temporary aeration basins. Figure 172b illustrates the latter scheme which is currently (1985) being evaluated in Pennsylvania.[95]

Air-Lift Pump. As Figure 172c shows, this approach combines air stripping with pumping. Early in the 20th century, air-lift pumps were commonly used in the United States. These pumps are simple devices with only two pipes in the well. One pipe introduces compressed air into the open bottom of the other pipe, called an eductor. In the eductor, air mixes with water. The mixture, being less dense than the surrounding water, rises. The air and the VOCs are separated before the water is pumped into the distribution system. Because of poor pumping efficiency, typically 35 percent, air-lift pumps fell into disuse with the introduction of submersible pumps. A utility in Pennsylvania is evaluating air-lift pumping to determine whether resurrecting this technology is less costly (trading poor efficiency for reduced capital construction) than other treatment alternatives.[95] A preliminary assessment showed 40 to 70 percent removal of trichloroethylene with A/W ratios below 10 to 1. This technology, therefore, may be sufficient for utilities needing only modest removal efficiencies to achieve their water-quality goals.

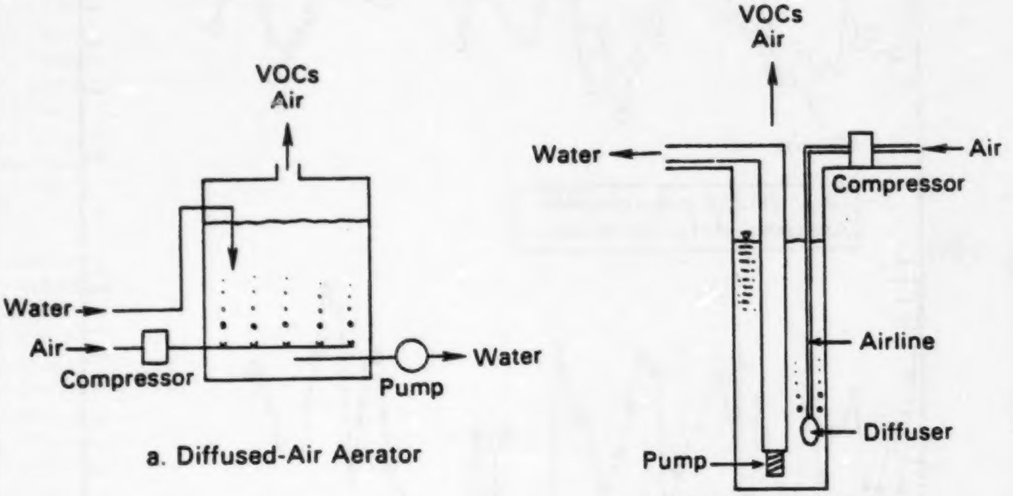
Mechanical Surface Aeration. Figure 172d illustrates this alternative. Like diffused-air aeration, mechanical surface aerators are commonly used in waste-water treatment. Roberts and Levy[96] see these devices as offering many advantages for air stripping of VOCs from drinking water. Several types are available. They can be mounted on platforms or bridges or supported by columns or pontoons. A motor-driven, impeller-like turbine creates turbulent mixing of air and water. These devices combine both features of putting air through water and, the next category, water through air.

Water-Through-Air Aeration Systems

Packed Tower Aeration. Approximately two-thirds of the aerators installed within the United States for VOC control are packed-tower aerators.[92] Figure 174a illustrates this device which consists of a column, typically 1 to 3m in diameter and 5 to 10 m in height, that is filled with packaging material. This packing can be glass, ceramic, or plastic and is available in numerous geometrical shapes to create a water film on a large surface area, thus, enhancing transfer of VOCs to the air phase. The inside wall of the aeration column has several redistributors that force the water to flow over the packing and thus prevent the water from simply running down the walls. Figure 174a shows a fan at the top; this device is called an induced-draft packed tower. Figure 174a also shows the more common forced-draft packed tower where a blower positioned at the bottom of the tower forces air up through the packing. Several excellent references[97–102] are available that describe these devices, discuss their theory and design, and present data that demonstrate 98 percent or more removal of the VOCs found in ground water. Reference 92 presents similar performance data for this type of aerator.

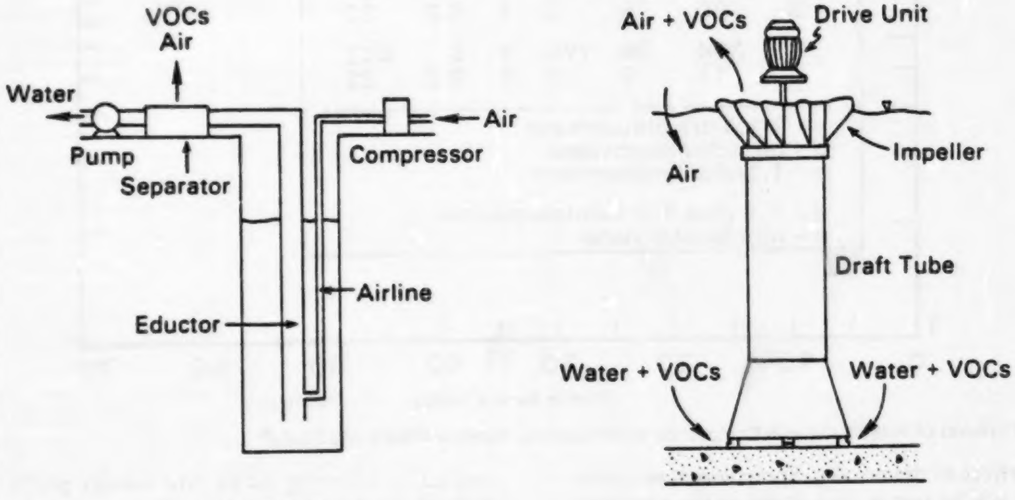
Tray Aeration. In a tray aerator, water is pumped to the top of the device, distributed over the cross-sectional area by a perforated plate, and allowed to trickle down and over trays which, in Figure 174b, are redwood slats. Water splashing on the slats, which are staggered to prevent the water from short circuiting the device, creates the air-water interface for mass transfer. Air is drawn into the tower by fans or blown in as described earlier for packed towers. The air travels upward, counter-current, to the water flow. These devices are commonly used for controlling inorganic compounds (iron, manganese, hydrogen sulfide, carbon dioxide) in drinking water and units have been modified (additional blower and slats) to treat VOCs.[103] Cooling towers are also included in this category. For cooling towers, however, Figure 174c shows that the air movement might be cross-current, rather than counter-current.

Spray and Venturi Draft Aeration. According to Cheremisinoff[104], these devices, illustrated in Figure 174d, evolved from spray ponds that were surrounded by louvered walls. By spraying water through nozzles, small droplets having a large interfacial surface area are produced. In a natural-draft spray tower, air movement is dependent on the atmospheric conditions and the



a. Diffused-Air Aerator

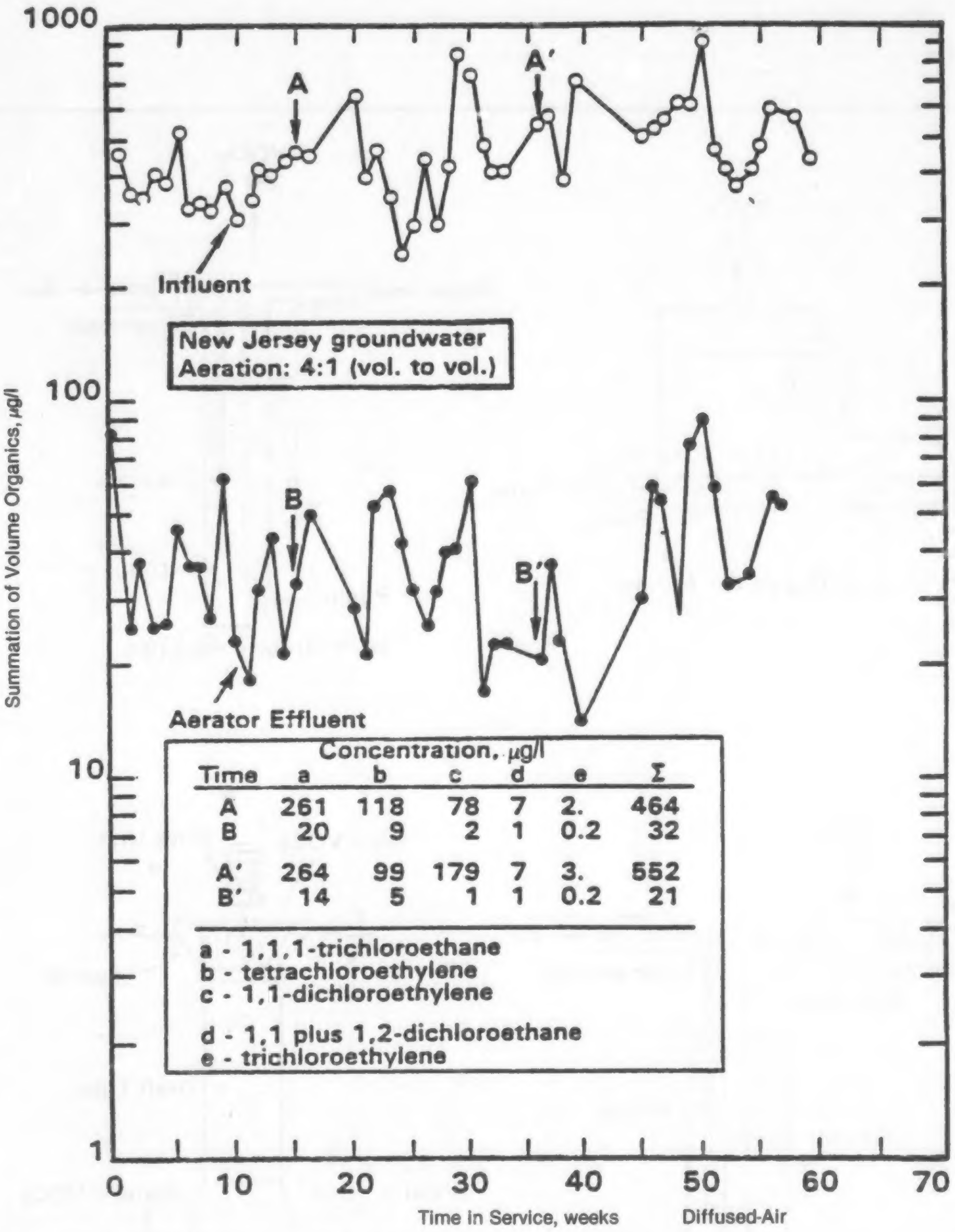
b. In-Ground Aeration



c. Air Lift Pump

d. Mechanical Surface Aerator

Figure 172. Aeration Devices that Put Air through Water



Concentration, μg/l

Time	a	b	c	d	e	Σ
A	261	118	78	7	2.	464
B	20	9	2	1	0.2	32
A′	264	99	179	7	3.	552
B′	14	5	1	1	0.2	21

a - 1,1,1-trichloroethane
b - tetrachloroethylene
c - 1,1-dichloroethylene

d - 1,1 plus 1,2-dichloroethane
e - trichloroethylene

Figure 173. Removal of Volatile Organic Compounds by Diffused-Air Aeration (Pilot-Scale Study)[94]

aspirating effect of the nozzles. Spray towers are generally placed side by side at right angles to the prevailing winds. In a natural-draft device air is induced as a result of water injected through spray nozzles into a venturi-shaped plenum. The Calgon Corporation* has incorporated one of these devices with an activated carbon adsorber to combine aeration with carbon adsorption.

Catenary Grid Scrubber®†. This is a proprietary device, illustrated in Figure 174e, that was developed as an air scrubber[105], but may have an application for removing VOCs from water. A fluidized air-water contact is created by balancing the air-flow velocity profile through catenary-shaped grids. These grids, generally placed in pairs, are stainless steel wire mesh with 0.6 to 2.5 cm openings. The manufacturer reports a very low air pressure drop and speculates that this device would be perhaps one-half the height of a packed tower.

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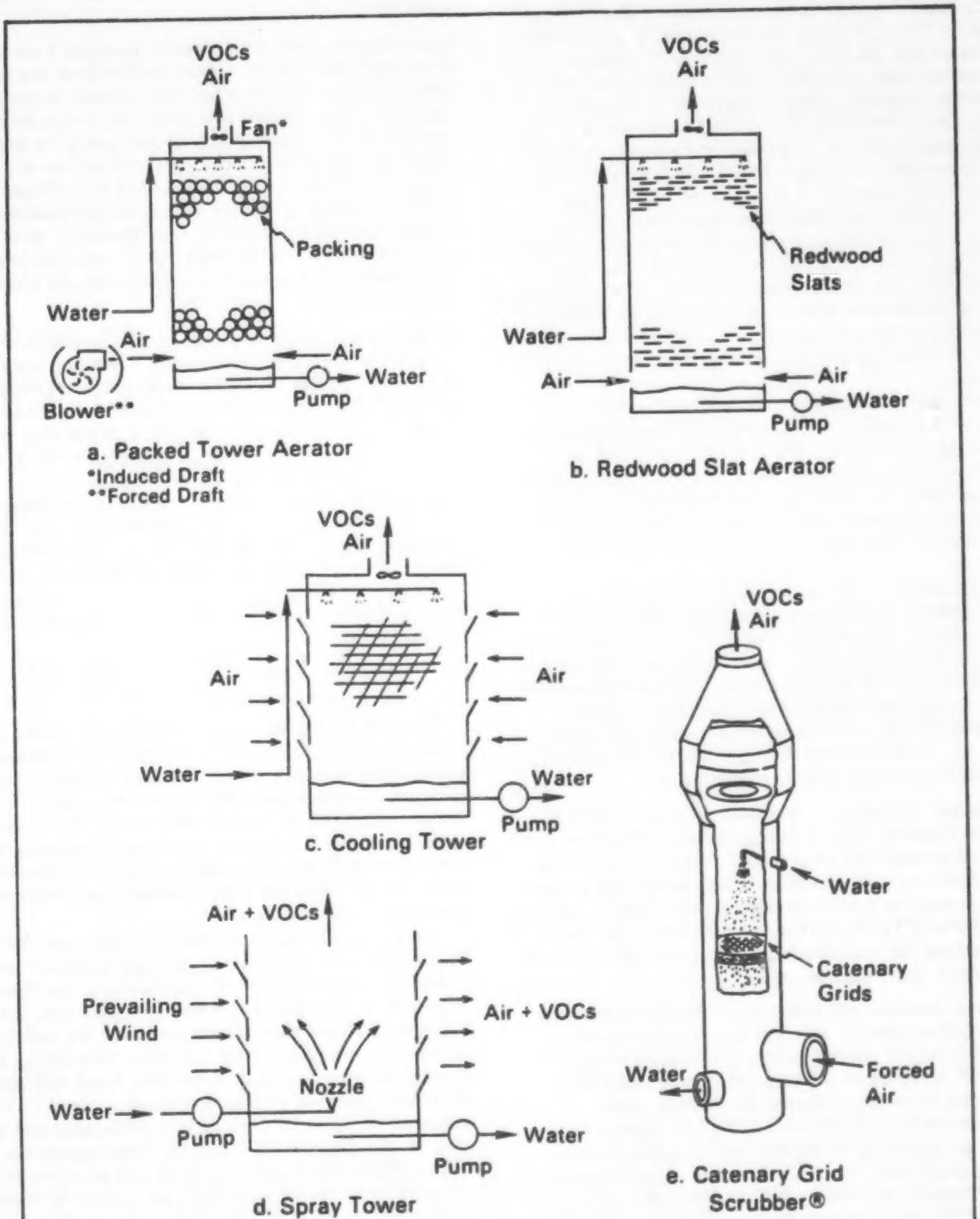
*Pittsburg, Pennsylvania. Interphase®Ground-Water Treatment System

†Chem-Pro Corporation, 17 Daniel Road, P.O. Box 1248, Fairfield, New Jersey



a. Packed Tower Aerator

*Induced Draft

**Forced Draft

b. Redwood Slat Aerator

c. Cooling Tower

d. Spray Tower

e. Catenary Grid Scrubber®

Figure 174. Aeration Devices that Put Water through Air

Operational Aspects (Secondary Effects) of Aeration

Air Emission. Aeration is not a technology that destroys or alters VOCs, it simply transfers them to the ambient air where they are dispersed, diluted and, perhaps, photochemically degraded. This is, perhaps, the most important secondary effect of aeration to consider.

An EPA-DWRD project in the Los Angeles basin is identifying the off-gas dispersion pattern, and quantifying, if possible, the VOC concentrations emitted from a packed tower aerator.[106] The state of California has rigid constraints on air emissions from point sources and a discharge permit was required before this aeration research project could proceed from pilot-scale operation to full-scale design. Umphres, *et al.*,[107] documented the effort that was required to receive the air discharge permit and discussed the state's (January 1983) requirement for "best available technology" for air pollution control.

The state of Michigan, requires "best available technology" be applied to VOC treatment devices discharging to the air. Further, "carcinogens" must not be transferred to ambient air. Thus, the state requires granular activated carbon (GAC) treatment of the off-gas stream from aerators used to remove VOCs from drinking water or from aerators used to remove VOCs from aquifers.

Designers must take care to ensure that state and local ambient air-quality standards are not exceeded in the vicinity of the aerator. The EPA-DWRD project in the Los Angeles basin has, as an objective, a laboratory examination of the gas-phase adsorption performance of GAC for off-gas treatment as a function of relative humidity.[106] The addition of off-gas treatment technologies, i.e., GAC and humidity control, can significantly increase the cost of aeration technologies.[108]

Particulates. Aerators are, in effect, large air scrubbers and the drinking water is in contact with 100,000 m^3 or more of air each day of operation. Ground-water treatment rarely includes filtration. Thus, an increase in particulates may be possible in an area with high air particulates. EPA-DWRD's contractors have not been able to demonstrate any measureable water-quality deterioration from particulates after aeration[92].

Corrosion. Aeration will strip carbon dioxide from water and introduce modest increases in pH. Additionally, dissolved oxygen concentrations will increase slightly.[92, 95, 96, 109] A survey of full-scale aerators revealed these conditions have not produced distribution system materials problem.[92] EPA-DWRD's Los Angeles project has as an objective, an examination of corrosion rates in aerator effluent waters[106]. One area of corrosion concern, however, lies with in-well aeration, shown in Figure 172b and c, where, because of the pressure head, dissolved oxygen concentrations can reach saturation[95] which may be corrosive in some distribution systems.

Bacteriological Quality. Increases in standard plate count and, occasionally, in total coliform densities, occur with aeration.[109] An aerator is a "break-in-the-system" and disinfection should be incorporated following treatment. *Legionella* species have not been isolated in aerated water.[109]

Operational Problems. Cold weather does not have a demonstrable effect on aerator performance but some utilities have elected to house their aerator to protect against ice damage should a pump fail during inclement weather. A far greater potential problem is the precipitation of iron on the packing material within an aerator. Fronk and Love[109] described a significant loss of VOC removal efficiency when iron precipitation clogged an aerator. These devices, therefore, should have an access port so the internals can be routinely inspected and cleaned. Manganese and copper can also precipitate following aeration.

Estimating Aeration Performance. Several investigators have reported the efficiency of aerating VOCs and some have attempted to relate molecular properties to performance. McCarty and coworkers[110] concluded Henry's Law Constant (H), or Henry's coefficient, is the most useful indicator of relative ease for air stripping VOC. H can be defined as the ratio of the concentrations of a volatile compound in air and in water at equilibrium. H is expressed either as atmospheres (atm), atm-m^3/mole, or dimensionless. By expressing H as dimensionless, the reciprocal, 1/H (called the partition coefficient) is the theoretical optimum air-to-water ratio for removing a volatile compound by air stripping. To express H as dimensionless:

multiply H expressed as atm by 0.00056 (5.6×10^{-4})
multiply H expressed as atm-m^3/mole by 42

Knowing H, organic compounds can be ranked as is shown in Figure 175. Figure 175 reveals to important points about aeration: (1) all VOCs are strippable—some are more so than others and (2) the use of aeration to remove one VOC will remove all other VOCs to some extent. Figure 175 does not show ease of stripping to be a function of VOC concentration; for VOC concentrations typical to ground waters, aeration performance is independent of concentration.

Aeration needs can be simplistically estimated knowing H. Trichloroethylene, for example, has a Henry's Law Constant of 0.48 at 20°C.[111] The partition coefficient, therefore, is 1/0.48 or 2.1. Thus, at equilibrium, 2.1 volumes of air are theoretically necessary for each volume of water to strip out trichloroethylene. In practice, the theoretical optimum air-to-water ratio is not sufficient for 99 percent removal. For the hypothetical blend of VOCs described in Table 16, the Henry's coefficients and stripping factors are given in Table 19. From experience, EPA-DWRD has found that the actual air-to-water ratio averages approximately 20 times the theoretical value for diffused-air aeration.[94] This, then, gives an estimate of aeration requirements. Table 20 illustrates the result of a selected air-to-water ratio on the control by diffused-air aeration of the Table 16 VOCs. At an air-to-water ratio of 20:1, tetrachloroethylene effluent levels will be approximately 2 $\mu g/l$; the others having lower Henry's coefficients, will not have reached 99 percent removal. At a ratio of 400:1, 1,2-dichloroethane removal will be 99 per-

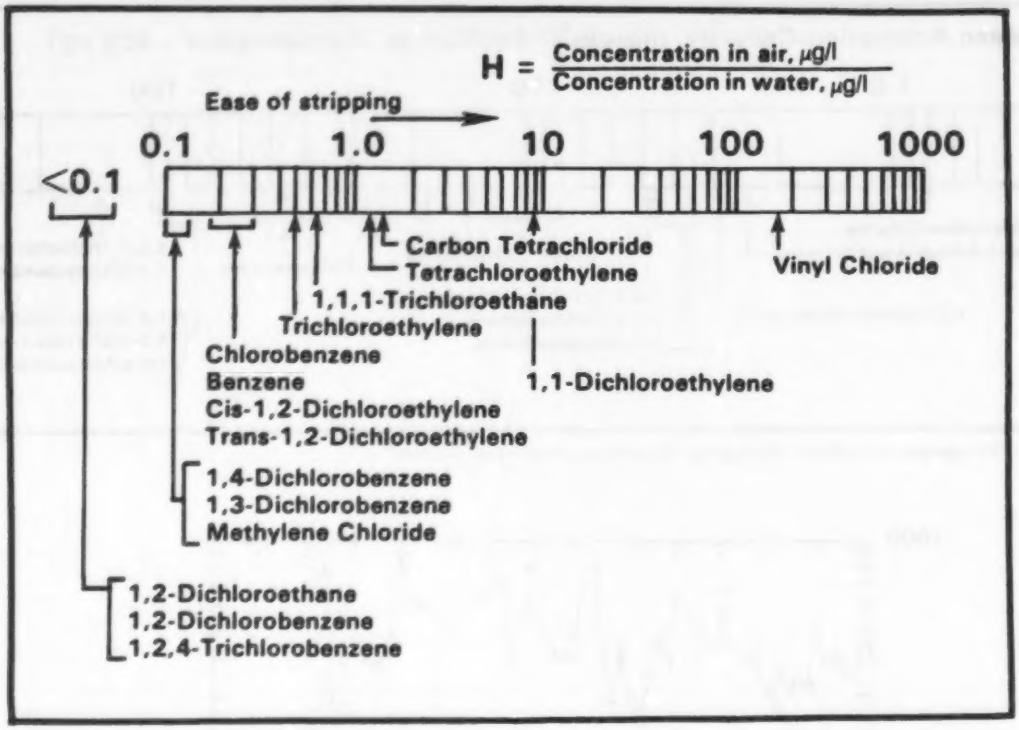


Figure 175. Comparison of Henry's Law Constants for Selected Organics[110,111]

cent; the others, having higher Henry's coefficients will have removals better than 99 percent. For packed-tower aeration, the actual-to-theoretical ratio is appreciably better, i.e., approximately 3. Recall that this exercise is a simple estimation of aeration needs. The importance of proper design based on pilot testing should not be underestimated. References 97 to 102 should be consulted.

Table 19. Estimating Aeration Needs for Diffused-Air Aeration

		99% Removal Air: Water Ratio	
VOC	H	Theoretical	Actual
Trichloroethylene	0.5	2	40
Tetrachloroethylene	1	1	20
cis-1,2-Dichloroethylene	0.3	3	60
1,2-Dichloroethane	0.05	20	400

Table 20. Water Quality Assuming 99 Percent Removal Efficiency by Diffused-Air Aeration (μg/L)

	Air:Water = 20:1	Air:Water = 400:1
Trichloroethylene	>5	<5
Tetrachloroethylene	2	<2
cis-1,2-Dichloroethylene	>1	<1
1,2-Dichloroethane	>0.1	0.1

Adsorption

Synthetic Resins. In 1976, the Rohm and Haas Company introduced an experimental synthetic resin called Ambersorb XE-340. This material was designed specifically for removing low molecular weight halogenated compounds because, at that time, chloroform and other trihalomethanes were the topic of intense research in the United States. Although this material looked very promising because of its high adsorption capacity, in 1982 the manufacturer announced that XE-340 would not be commercially produced. The future of resins for controlling VOCs is therefore questionable.

Powdered Activated Carbon. Although powdered activated carbon (PAC) is rarely employed in groundwater treatment where clarification is not required, it has been monitored in surface-water plants treating for VOCs. At the doses at which PAC can be applied, typically for taste and odor control, it has not been found to be cost-effective.[94]

Granular Activated Carbon. Figure 176, a compilation of data from several laboratory studies with activated carbon, illustrates two important points: (1) the organic compounds found in ground water cover a wide range of adsorption capacities and (2) treatment by adsorption to remove one VOC will removal all VOCs to some extent. In contrast to aeration, where performance was independent of concentration, the adsorption capacity varies directly with the equilibrium concentration of the organic contaminant.

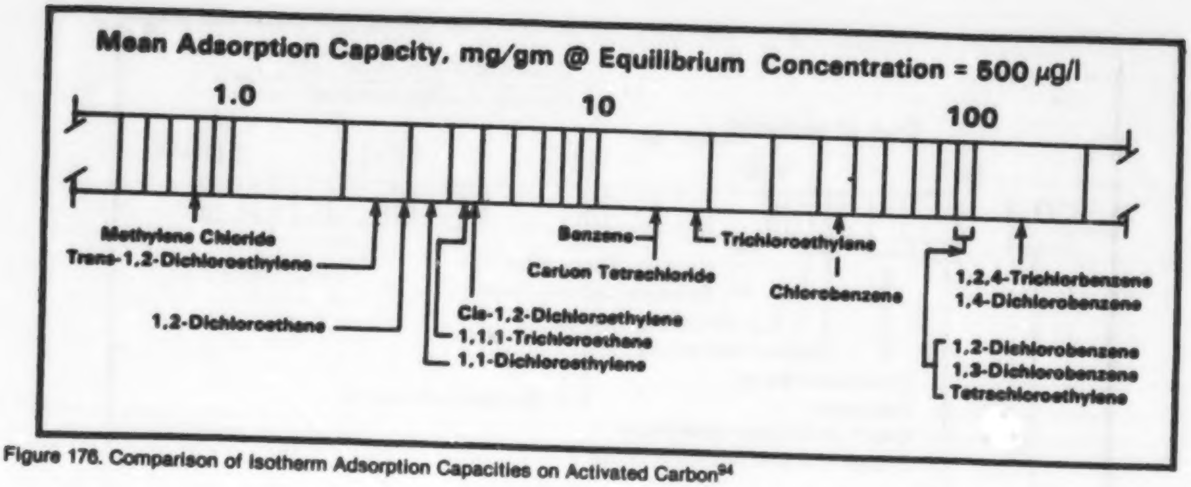
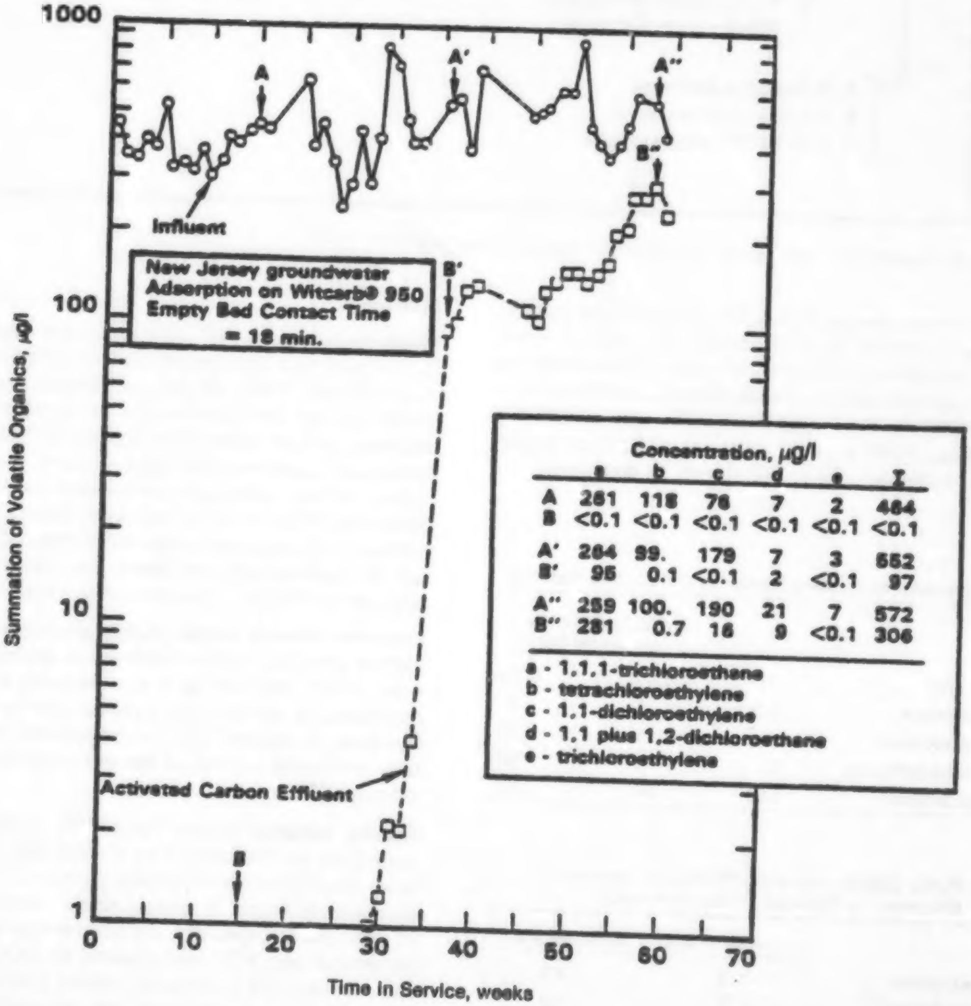


Figure 176. Comparison of Isotherm Adsorption Capacities on Activated Carbon[94]



Concentration, µg/l

	a	b	c	d	e	Σ
A	281	118	76	7	2	484
B	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
A'	284	99.	179	7	3	552
B'	95	0.1	<0.1	2	<0.1	97
A''	259	100.	190	21	7	572
B''	281	0.7	16	9	<0.1	306

a - 1,1,1-trichloroethane
b - tetrachloroethylene
c - 1,1-dichloroethylene
d - 1,1 plus 1,2-dichloroethane
e - trichloroethylene

Figure 177. Removal of Volatile Organic Compounds by Adsorption on Granular Activated Carbon (Pilot-Scale Study)[94]

Figure 177 summarizes data collected from an EPA-DWRD pilot-scale adsorption study. For approximately 30 weeks (17,000 bed volumes), the contaminants were removed to "below detection" (0.1 micrograms per liter, μg/l) by the GAC. This is typical of adsorption systems and is unlike aeration systems, as Figure 173 shows. When breakthrough occurred, the principal organic compounds were the chloroethanes, which is consistent with the order of adsorption capacities given in Figure 176. At point B″ the 1,1,1-trichloroethane concentration in the effluent slightly exceeded the influent indicating the carbon was exhausted for control of the compound.

Adsorption can be affected by the amount of background organic carbon, generally measured as TOC. High background organic content (i.e., over 1.5 to 2 milligrams per liter) can affect adsorption capacities. Competitive adsorption and the influence of high concentrations of background organic carbon are recognized, but not well understood. Table 21 demonstrates how, in one study, increasing TOC decreased the adsorption capacity of trichloroethylene[112]. This again demonstrates the importance of defining TOC.

Table 21. Effects of Competitive Adsorption on Trichloroethylene[112]

TOC mg/l	Freundlich Isotherm Coefficients	
	1/n	K (Capacity at 1.0 mg/l) mg/gram
<0.1	0.48	56.0
1.97	0.44	31.6
8.39	0.44	24.4

Operational Issues of Adsorption

Spent Adsorbent. At some point following breakthrough, effluent concentrations of a contaminant will exceed effluent quality levels and the carbon must be replaced or reactivated. Typically ground-water utilities have not employed onsite reactivation, but have employed an adsorption service. Two manufacturers of activated carbon (Calgon Corporation of Pittsburgh, Pennsylvania, and ICI Americas of Wilmington, Delaware) offer adsorption services and approximately a dozen water utilities have employed this treatment option. In general, the service consists of characterizing the water, sizing, constructing and installing pressure adsorption vessels, and replacing the spent GAC with virgin GAC material when the concentration of a contaminant exceeds a predetermined effluent quality. The adsorbent is then returned to the manufacturer, reactivated, and resold for industrial use. These adsorbers typically contain up to 9,000 kg (20,000 lb) of activated carbon. Utilities not employing an adsorption service must plan for disposal of the spent GAC. One firm (Carbon Services of Minneapolis, Minnesota) offers gravity adsorbers that hold approximately 2,500 kg (6,000 lb) of activated carbon. These adsorbers are off-the-shelf refuse dumpsters, that have been lined for corrosion protection and can be conveniently hauled onsite, leveled, and put into service. When effluent quality concentration has been exceeded, the dumpsters can be lifted onto a truck and the GAC transported for disposal, reactivation, or replacement.

Pressure adsorbers are normally operated in series with empty bed contact times typically ranging from 8 to 18 minutes. The time of operation varies because at this writing (1985), no national maximum contaminant levels (MCLs) exist and the action levels of the various states are not uniform. Pressure adsorbers are typically designed to operate 8 or more months. Operating in series allow each adsorption vessel to remain on-line until the activated carbon is fully exhausted before it is replaced. This two-stage, series operation and the steps involved are shown in Figure 178. In contrast to adsorbers operating in parallel or alone, the series operation uses a significantly larger portion of the adsorbent.

Desorption. When an adsorbant is exhausted, a decrease in the influent concentration will upset the equilibrium between the adsorbed contaminant and the contaminant in solution. Under these conditions, or whenever there is a significant decrease in the influent concentration, desorption can occur. An example of this concentration is seen at point B″ in Figure 177 for 1,1,1-trichloroethane. In addition to equilibrium, desorption can occur as a result of displacement by a more competitive compound. It is important to sample not only the GAC effluent to monitor VOC concentrations reaching the consumer, but also the GAC influent to better understand GAC contactor behavior.

Bacteriological Quality. Increases in standard plate count and total coliform densities are well known across GAC contactors treating surface waters, especially in summer months when water temperatures rise. GAC will eliminate the free chlorine residual and adsorb nutrients onto its surface. While bacterial densities in ground waters are generally several logs lower than surface waters and while ground-water temperatures are relatively constant, bacterial densities can increase across GAC contactors treating ground water. Thus, a GAC contactor should be followed by a disinfection process prior to water distribution.

Estimating Adsorption Performance. Certain molecular properties may be useful for preliminary estimates of adsorption capacity when confronted with a newly identified contaminant. Love and Miltner[113] correlated adsorption capacity with molecular weight, density, vapor pressure, solubility, Henry's Law Constant, parachor, and molar refraction. The parachor relates molar volume (ratio of molecular weight to density) to surface tension. The parachor of a compound can be estimated from the sum of the atomic and structural parachor equivalents given in Table 22.

Table 22. Atomic and Structural Parachor Equivalents[114]

Carbon	4.8	Chlorine	53.8	Carbon double bond	23.2
Hydrogen	17.1	Bromine	68.0	Carbon triple bond	46.6
Oxygen	20.0	Iodine	91.0	Aromatic ring	6.1

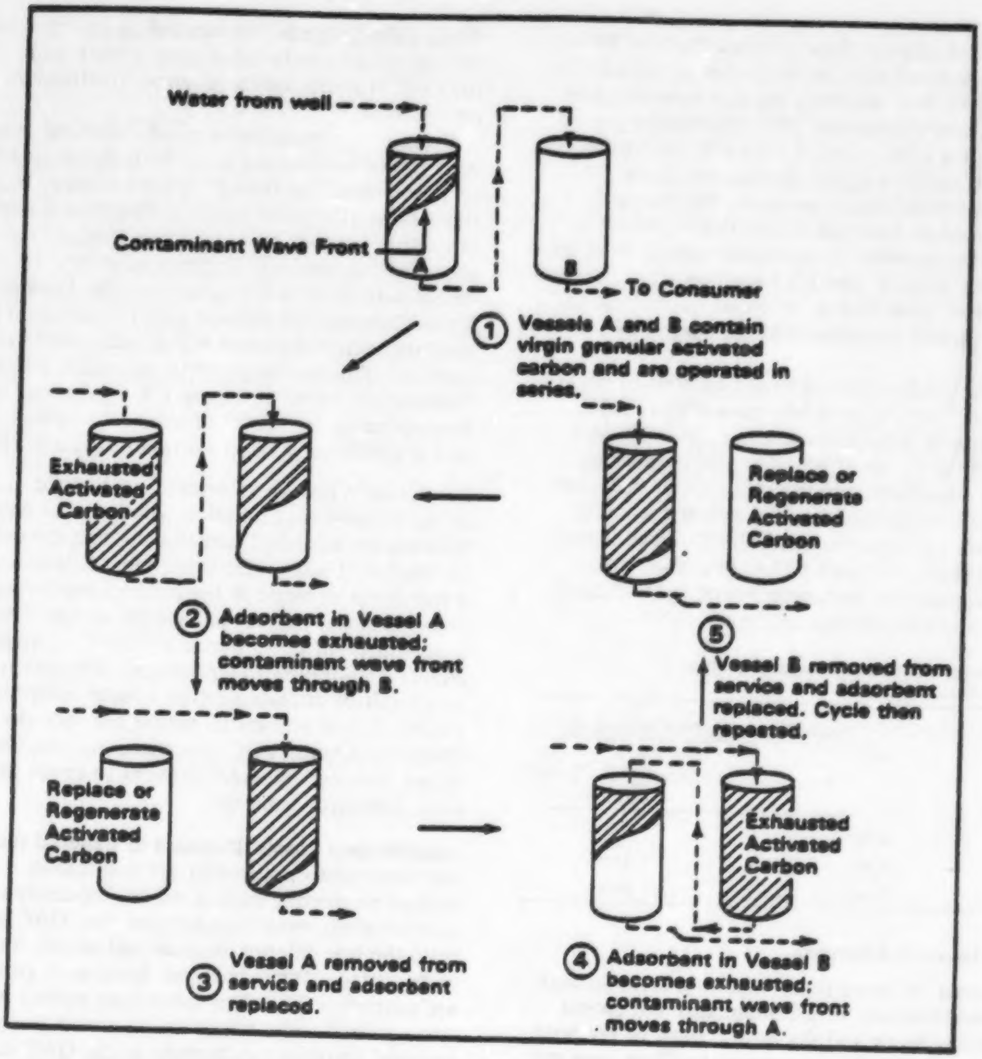

Water from well
Contaminant Wave Front
A
B
To Consumer
① Vessels A and B contain virgin granular activated carbon and are operated in series.
Exhausted Activated Carbon
② Adsorbent in Vessel A becomes exhausted; contaminant wave front moves through B.
Replace or Regenerate Activated Carbon
③ Vessel A removed from service and adsorbent replaced.
Exhausted Activated Carbon
④ Adsorbent in Vessel B becomes exhausted; contaminant wave front moves through A.
Replace or Regenerate Activated Carbon
⑤ Vessel B removed from service and adsorbent replaced. Cycle then repeated.

Figure 178. Two-Stage Series Operation to Maximize Use of Activated Carbon

The molar refraction of a compound describes the relationship between molar volume and the compound's refractive index. Like the parachor, molar refraction can be approximated by adding the appropriate atomic and structural refraction equivalents shown in Table 23.

Table 23. Atomic and Structural Refraction Equivalents[114]

Carbon	2.42 ml	Carbon double bond	1.73 ml
Hydrogen	1.10	Carbon triple bond	2.40
Chlorine	5.97	Carbonyl oxygen	2.21
Bromine	8.87	Hydroxyl oxygen	1.53
Iodine	13.90	Ether oxygen	1.64

Molecular weight, density, and Henry's Law Constant produced poor correlations with adsorption. In Figure 179, three *arbitrary* ranges of adsorption capacity (poor = < 10 mg/g; fair = 10–50 mg/g; good = > 50 mg/g) are shown with corresponding fitted curves ($y = bx^c$) for solubility, vapor pressure, molar refraction, and parachor. Assume, for example, that a newly identified organic compound is known to have a low solubility and vapor pressure, and the parachor and molar refraction are calculated to be relatively high. That organic compound is likely to be well adsorbed on granular activated carbon. This approach is not intended to replace gathering the actual adsorption data, but might be useful in obtaining preliminary information for adsorption capacity where empirical data are lacking.[113]

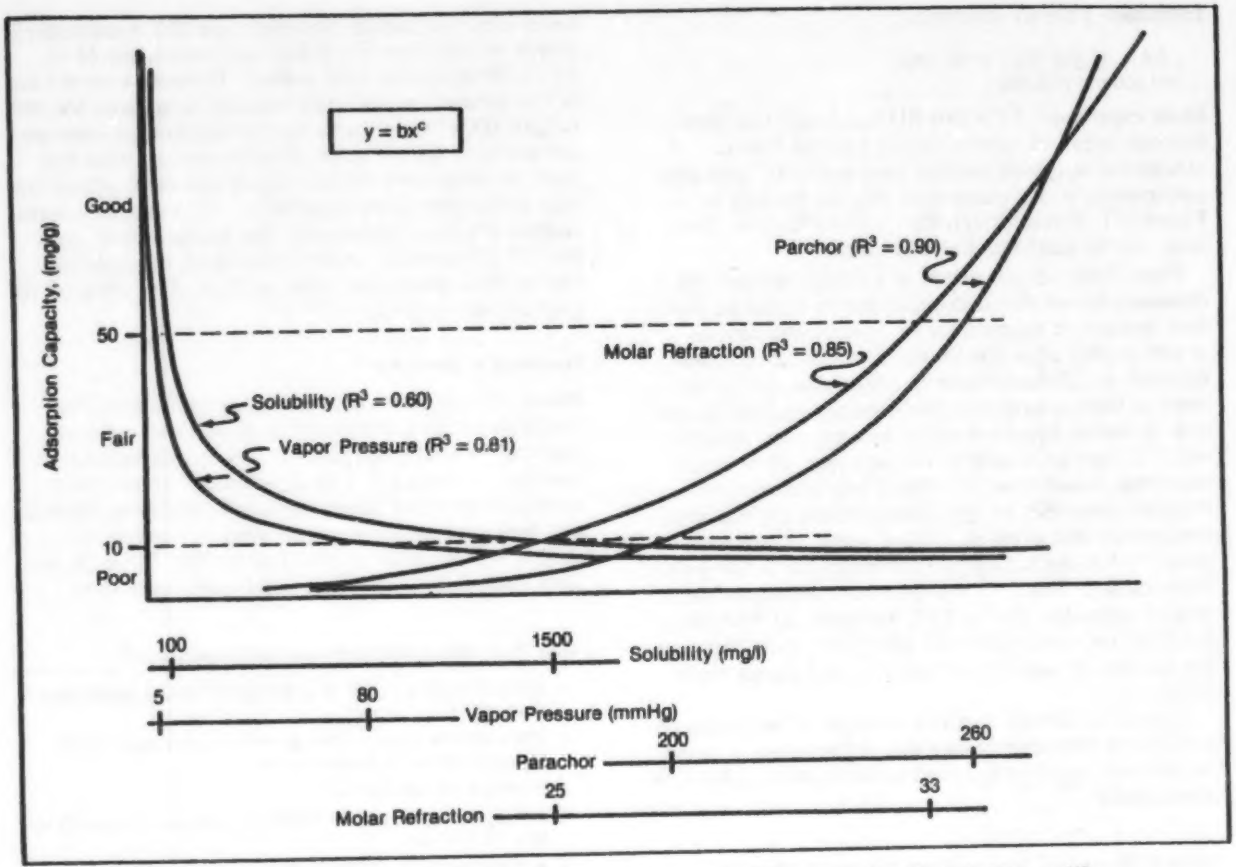


Figure 179. Use of Molecular Properties to Estimate Adsorption of Volatile Organic Compounds on Activated Carbon[113]

Adsorption performance can be roughly estimated starting with the Freundlich equation:

$$\frac{X}{M} = KC_e^{1/n} \qquad (62)$$

where:

X/M = adsorption capacity, mg VOC adsorbed/gram activated carbon

C_e = equilibrium concentration of the VOC, mg/l

K and 1/n are empirical constants characteristics of compound, the water, and the activated carbon.

The K and 1/n values are published for certain compounds[112, 113, 115] or can be obtained through laboratory adsorption isotherm studies. Some of these data give the K and 1/n values for trichloroethylene as 36 mg/g and 0.5, respectively, in distilled water. Therefore, at an equilibrium concentration (C_e) of 500 µg/l (0.5 mg/l) the adsorption capacity would equal:

$$x/M = KC_e^{1/n} = 36(0.5)^{0.5} = 26 \text{ mg/g}$$

Similarly, the adsorption capacities for the VOCs in Table 16 are shown in Table 24.

Table 24. Adsorption Capacity Based on Isotherm Data

VOC	K, mg/g	1/n	X/M, mg/g
Trichloroethylene (TCE)	36	0.5	26
Tetrachloroethylene	134	0.5	60
cis-1,2-Dichloroethylene	8	0.5	3
1,2-Dichloroethane	5	0.6	0.4

Assuming a well with this blend of VOCs is pumped continuously at 200 gal/min, and that 20,000 pounds of granular activated carbon are to be used, the estimated time to exhaustion may be determined:

Estimated TCE loading per day:

$$200\ \frac{\text{gal}}{\text{min}} \times 3.78\ \frac{\text{liter}}{\text{gal}} \times 1{,}440\ \frac{\text{min}}{\text{day}} \times \frac{0.5\ \text{mg TCE}}{\text{liter}} = 544{,}300\ \frac{\text{mg}}{\text{day}}$$

Estimated Adsorption capacity:

$$20{,}000\ \text{lb GAC} \times 454\ \frac{\text{g}}{\text{lb}} \times 26\ \text{mg}\ \frac{\text{TCE}}{\text{g GAC}} = 2.4 \times 10^8\ \text{mg TCE}$$

Estimated Time to exhaustion:

$$\frac{2.4 \times 10^8 \text{ mg TCE}}{544{,}300 \text{ mg TCE/day}} = 430 \text{ days}$$

From experience, EPA-DWRD has found that breakthrough occurs at approximately half the time to exhaustion at typical loading rates and GAC contactor configurations. An example of this can be seen in Figure 177. Breakthrough for trichloroethylene, therefore, can be estimated to be 215 days.

From Table 25, calculated in a similar manner, the estimated life of this adsorption system would be about four months, if reactivation or replacement were required shortly after breakthrough of cis-1,2-dichloroethylene or 1,2-dichloroethane. Note that the contaminant in highest concentration does not necessarily control. A similar approach might be used if the situation was: "I want to be able to run one year. How much adsorbent would I need?" Recall that adsorption isotherm capacities are generally reported on individual compounds and often in distilled water, so the background total organic carbon concentration in the ground water must be low (<1 mg/l) to have much confidence in such estimates. As the TOC increases, so does the potential for competition for adsorption sites; therefore, the adsorption capacity for specific compounds might decline.

It must be stressed that this example of estimating adsorbtion performance is, like that for aeration, a simplification and *must* be followed up with onsite, pilot-scale investigation.

Table 25. Estimated Time for Breakthrough on Granular Activated Carbon

VOC	Days
Trichloroethylene	215
Tetrachloroethylene	1,430
cis-1,2-Dichloroethylene	130
1,2-Dichloroethane	110

Combination of Treatment Processes. Two or three water utilities in the United States have preceeded adsorption with aeration and have found this combination to be extremely effective for producing a water with volatile organic concentrations below a detection limit of 0.1 µg/l. The aeration step reduces the organic load to the adsorbent and may remove compounds competing for adsorption sites. The Calgon Corporation has a process called Interphase® that incorporates a venturi-draft aerator with a gravity-flow adsorber.

Relative Costs of Treatment. The extent of treatment provided will be governed by the desired effluent quality and the associated costs. Cost estimates vary significantly because each process has a spectrum of effectiveness. From the available estimates[116, 117] some generalizations can be made on relative costs. Packed-tower aeration is considerably less expensive than diffused-air aeration or adsorption on activated carbon. The cost of packed-tower aeration (capital plus operation and maintenance) ranges from 0.03 to 0.1 dollars per cubic meter (4 to $13^{¢}$/1,000 gallons in 1982 dollars). Depending on the size of the aerator, capital costs typically range from $60,000 to $200,000 (1982 dollars). Similar cost comparisons are not available for the other aeration devices. Note that these aeration costs do not include any expenditures that may result from secondary effects. For example, off-gas control has been estimated to double the cost of aeration.[108] Adsorption, on the other hand, typically costs two to three times more than aeration, depending on the contaminant of concern.[108, 117]

Treatment in the Home

Boiling. One of the first questions generally asked by health department personnel or homeowners after discovering VOC contamination is: "Can the organics be removed by boiling?" This is probably because historically contaminated water contained pathogenic bacteria and boil-water orders come to mind as the first course of action. VOCs can be removed by boiling. Table 26 contains a summary of relevant points and precautions.

Table 26. Boiling to Remove Volatile Contaminants[94]

- All have boiling points or azeotrophic boiling points less than 100°C.
- The depth of water in the pan affects efficiency; better removals occur in shallow pans.
- Boiling must be vigorous.
- A timer is necessary; 5 minutes is sufficient to give 95 to 99+% removal.
- A range hood should be used to vent the VOCs out of doors.
- Inorganics will be concentrated.
- VOC inhalation exposure from showering will not be eliminated.

Point-of-Use Adsorption Units. In some circumstances, such as where a central treatment system is not feasible, or in emergency situations, point-of-use treatment devides using activated carbon adsorption have been successful.[118] Point-of-use devices which use varying amounts of adsorbents are available. The smallest units attach to the spigot and provide little contact time. Larger units treat water prior to the spigot. Reference 119 summarizes studies of microbial levels across point-of-use devices—primarily of the larger size. Finally, whole-house units are available. The state of New York[120] recommends only whole-house units because of the possibility of inhalation exposure to VOCs during and after showers in the confines of a bathroom. The exposure issue has been studied[121] but is still a debated topic.

A plumbing problem that must be avoided when installing and operating whole-house units is the introduction of untreated water when the adsorption units are being backwashed. This permits adsorption of organics in the bottom of the activated carbon bed. When returned to service, the organics desorb into the treated water. Perhaps a more significant problem with point-of-use devices is insuring that they receive the care necessary to maintain a desirable water quality. The National Sanitation Foundation[122] has evaluated the surveillance problems related to providing the maintenance and attention these units require.

Summary

Organic solvents of industrial origin occur in groundwater supplies and both non-treatment and treatment actions are being taken by water utilities. Aeration, adsorption on granular activated carbon, or a combination of these processes are the technologies of choice. The following steps are suggested when faced with an organics problem:

1. Characterize the Problem. Attempt to locate and stop the source of VOC contamination making certain the problem is not from materials used to treat, transport, or store the drinking water. Each well in the system should be sampled, as should several locations in the distribution system. If possible, conduct a hydrogeologic investigation.

2. Analyze the Data. The background total organic carbon (TOC) and total organic halogen (TOX) concentrations are important supplements to knowing specific organic concentrations. Be certain that the VOCs have been properly identified.

3. Solve the Problem. What are the non-treatment alternatives? What are the treatment alternatives? What are the budget constraints? What finished water quality is desired or required? Be sure the solution does not substitute for the organic problem a situation of microbiological or inorganic deterioration, or increased corrosion potential, or of prohibitive air emissions.

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Chapter 8

Inground Treatment, Restoration, and Reclamation

Prevention of ground-water contamination is far more logical, simple, and cost effective than attempting to correct a problem—a problem that may have been in existence for years. A great deal of time, effort, and money are presently being expended to develop remedial measures to counteract the effects of contaminated aquifers and public water supplies. These include traditional as well as innovative construction techniques, water management, and research initiatives. As far as the public or private water purveyor is concerned, however, only a few of these methods are within their realm of possibility, owing largely to cost considerations.

Assuming, beyond a reasonable doubt, that the ground-water supply system is actually contaminated, one has several options: (1) provide inground treatment/containment, (2) provide above ground treatment, (3) remove or isolate the source of contamination, (4) abandon the source of supply, or (5) ignore the problem. Generally, several techniques are coupled in order to achieve the desired results. None of the options, however, are entirely satisfactory, but in order to maintain public confidence, something generally will need to be done and, most likely, in an open manner.

Inground Treatment/Containment

In some situations it might be reasonable to attempt to degrade or immobilize a contaminant or to contain it within a specified or general area. In this manner, uncontaminated wells could still be operated but pumping schedules and rates would need to be maintained in such a way that the cones of depression would not interfere with the containment. Inground treatment/containment methods appear attractive in two widely different situations: (1) where the contaminant covers a relatively small area and can be immobilized and (2) where contaminants cover a broad area in which the surrounding strata are amenable to the emplacement of barrier walls or the installation of discharge or recharge wells.

In Situ Encapsulation. This technique has proven successful in areas where the contaminant can be immobilized chemically or by means of a grout. One interesting example, reported by Williams,[123] involved a 4,200 gallon leak of acrylate monomer from a corroded pipeline at a small plant in Ohio. The contaminant migrated through a layer of fill, consisting largely of cinders, and then downward through a storm sewer trench into a thin sand and gravel aquifer. A test boring and soil sampling program delineated the plume and indicated that the contaminant was slowly beginning to undergo polymerization and, therefore, immobilization. To increase the rate of reaction, 2-inch diameter perforated PVC pipe was buried, about 2 feet below land surface, in four narrow trenches that trended across the plume. A riser and manifold header connected each pipe to solution tanks containing a catalyst in one and an activator in the other. Both solutions contained a wetting agent. A total of 8,000 gallons of solution were injected during the two treatment operations and 1,000 gallons had been injected previously during the investigative phase. On the basis of pre- and post-treatment soil borings, it was estimated that 85 to 90 percent of the liquid monomer contaminant was solidified and in some places it exceeded 99 percent polymerization. It is assumed that the remaining material will polymerize naturally.

Depending on the hydrogeologic situation and the chemical characteristics of the waste, it would appear that similar techniques could be used elsewhere. Where chemical immobilization can not be achieved, it might be possible to grout in order to limit the potential for leaching and movement.

Microbial Degradation. Many, if not most, organic compounds can be degraded, at least to some extent, by microorganisms that occur naturally in the subsurface. Aquifer restoration by means of microbial degradation is still in its infancy, techniques are crude at best, and most results have not been entirely satisfactory. In general, microbial restoration techniques revolve around the injection of oxygen and inorganic nutrients, such as nitrogen and phosphorus sources, into the subsurface in order to encourage population increases of the indigenous microbial flora. Great numbers of microbes occur naturally underground, but apparently in reduced numbers because of limitations brought about by the scanty food supply. Introduction of outside sources of energy may cause rapid increases in the population and these, in turn, can degrade a fairly wide range of organic compounds. When the external source of energy is removed, the population will die back.

Several problems exist with the concept of microbial restoration. First, there must be available microorganisms that are compatible with the waste; that is, they

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must be able to degrade the waste so that it can be used as a source of carbon. Second, variations in concentration of the waste may kill or starve the organisms. Third, methods to evenly distribute the nutrients and oxygen, particularly where subsurface conditions are complex, are not well advanced. Furthermore, considerable care and monitoring are required to ensure that the injected air does not force the contaminant to migrate away from the zone of treatment and further compound the problem and to ensure that the degradation products that result from transformation are not more toxic than the original compound. Additionally, there appears to be a practical limit, both technically and economically, to the degree of restoration. Attempting to reduce the concentration of the contaminants to a zero level would likely required expenditures of both time and money that far exceed the benefits derived.

Fortunately, a great deal of research is presently being conducted on the role and potential use of microorganisms in aquifer restoration. Early results appear promising. On the other side of the coin, however, this technology is so new, the data base so small, and our ignorance so large that it will probably require many years before these methods become routine. The path between laboratory experiment and successful field implementation is generally tortuous, filled with blind alleys, and rutted with bad experiences and lack of practicality.

Barrier Walls. The emplacement of a wall of very low permeability across the path of a contaminant plume, around a contaminated source or area, or around an uncontaminated area, has been used with considerable success for several years. Barrier or cutoff walls, most commonly, are used in conjunction with well systems. The purpose of the wells is to withdraw the contaminated fluid so that it can be treated and, perhaps, reinjected. Obviously, the construction of barrier walls is expensive, but perhaps the most important consideration is the long-term cost of pumping and water treatment, which may require decades.

Barrier walls generally are constructed by one of three different methods: (1) slurry-wall, (2) grout curtain, or (3) sheet piling. In the slurry-wall system a trench is excavated in the proper location and filled with a mixture of bentonite and soil or, perhaps, cement or special additives. The foundation should lie on, or preferably in, an underlying unit of low permeability so that contaminants do not flow under the wall. Slurry walls, under proper conditions, can be constructed to depths of 100 feet or so.

Grout curtains are formed by driving or placing grouting pipe in drilled holes that extend downward to a unit of low permeability. Cement or chemical grout is injected in a number of primary holes under high pressure as the pipe is withdrawn. The next set of holes are placed midway between the primary injection points and the process is continued until the area will take no more grout.

Sheet pile cutoff walls have been used for many years for excavation bracing and dewatering. Where conditions are favorable, depths of 100 feet or more can be achieved. The system consists of driving interlocking sheet piles down to a unit of low permeability. Unfortunately, sheet piling is seldom water tight and individual plates can move laterally several to several tens of feet when they are being driven. Acidic or alkaline solutions, as well as some organic compounds, can reduce the expected life of the system.

Two separate slurry walls have been constructed along parts of the margin of the Rocky Mountain Arsenal near Denver in order to contain plumes that originate on the plant property.[124, 125, 126] Along the north boundary, where unconsolidated sand and gravel occur with a thickness that averages about 30 feet, the slurry wall, about 2 feet thick, is 6,800 feet long. On the upgradient side are a series of 35 12-inch diameter discharging wells on 200 foot centers that pump contaminated ground water into a treatment facility. After flowing through a carbon filtration system the water is then reinjected into 50 6-inch diameter recharge wells on 100 foot centers on the opposite side of the barrier. Along the northwest boundary is a recently completed bentonite slurry barrier that is 1,425 feet long, extending southwestward from a bedrock high. The wall, excavated with the bentonite slurry trench method, is 30 inches wide and extends 3 feet into the underlying bedrock. The barrier contains about 7,000 cubic yards of backfill, obtained from a borrow pit and blended with the bentonite prior to emplacement. The barrier was constructed where the saturated thickness of the permeable material is less than 10 feet. Paralleling the downgradient side of the barrier is a series of 21 recharge wells, stretching nearly 2,100 feet along the Arsenal boundary. Directly behind the barrier and extending into the thicker part of the surficial aquifer are 15 discharge wells. The contaminated ground Water is pumped to a treatment plant and then reinjected into the recharge wells, thus forming a hydraulic barrier. Farther southeast along the boundary is another hydraulic barrier system, about 1,500 feet long, that consists of two parallel rows of discharge wells with 15 wells per row and, downgradient, a row of 14 recharge wells. The contaminanted water, originating from a spill, is pumped, treated, and then reinjected. This system and the one along the north boundary was put into operation in late 1981 and the system along the northwest boundary began operation in 1984. The cost of the construction of the systems as well as the operating expenses are unknown.

A modified barrier design was described by Arlotta and others.[127] Called Envirowall, this system consists of a trench that is lined with 100 mil high density polyethylene and backfilled with sand. It is installed with the slurry trench construction method. The system was tested in New Brunswick, New Jersey, in the fall of 1982.

Brunsing and Cleary[128] described a method of complete isolation by slurry-induced ground displacement or block displacement. Demonstrated in Whitehouse, Florida, a slurry wall was constructed around a small area, 60 feet in diameter, to a depth of 23 feet in unconsolidated material. Injection wells were then used to force a soil bentonite slurry outward along the bottom of the cell. Subsequent test holes indicated that the new floor of the cell contained 5 to 12 inches of slurry. In a case such as this, wells and a treatment system would be required to keep leachate from overflowing the cell unless an effective cover that totally prohibited infiltration was placed over the unit.

Cutoff Trenches and Ground-Water Dams. A cutoff trench is merely a drain that extends a sufficient distance below the water table to intercept contaminated ground water. Construction can be simple to complex depending on soil and subsurface conditions and on the thickness of the unsaturated zone. Since a cutoff trench is designed to serve as a ground-water discharge line, in a manner similar to a perennial stream, hydrologic and topographic conditions are critical to its success. That is, it can be effective where the water table is near land surface, the contaminants are moving at a relatively shallow depth, and the geologic conditions permit excavation. Cutoff trenches are effective in both high and low permeability materials, although the latter would likely provide less water, which in turn would require less storage and treatment costs. Furthermore, they are particularly suited to immiscible contaminants that tend to float. The trenches can remain open, filled with gravel, or be fitted with perforated underdrain pipes.

A ground-water dam installation was described by Giddings.[129] In this case, a landfill that began as a burning dump, was found to be discharging leachate both to the surface and the ground water, much of which eventually flowed into an adjacent river. A leachate interceptor trench was constructed on the downgradient side of the disposal area as shown in Figure 180. In the trench on the upgradient side was placed a perforated pipe in a gravel envelope that was covered with permeable material. The remainder of the trench on the downgradient side was then backfilled with fine-grained material as shown in Figure 181. Leachate from the landfill flows into the filled trench, seeps into the perforated pipe, and then is pumped back into the landfill. An application to discharge to a regional sewer line was under consideration at the time Giddings' report was published. In this case, the main

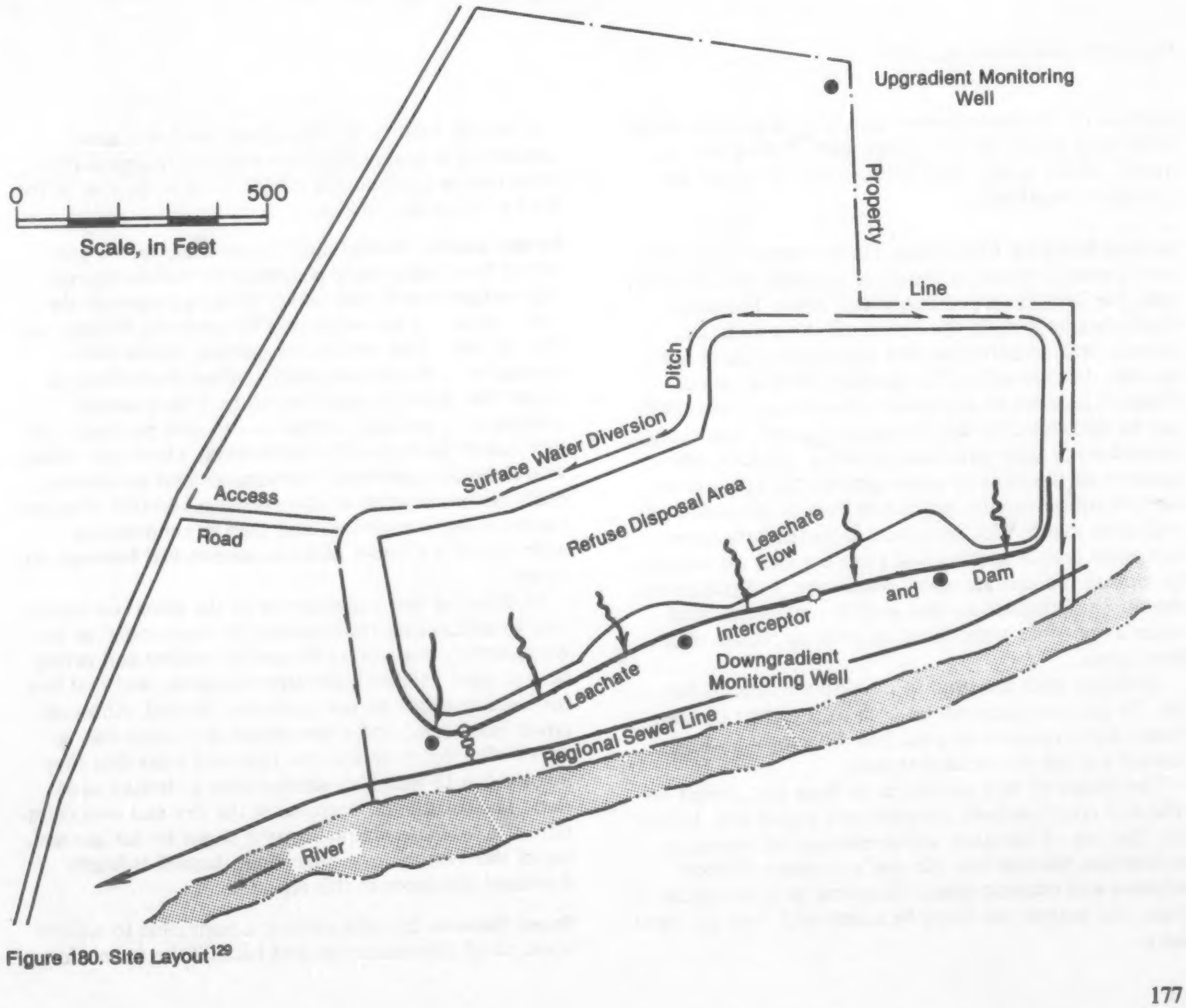

Figure 180. Site Layout[129]

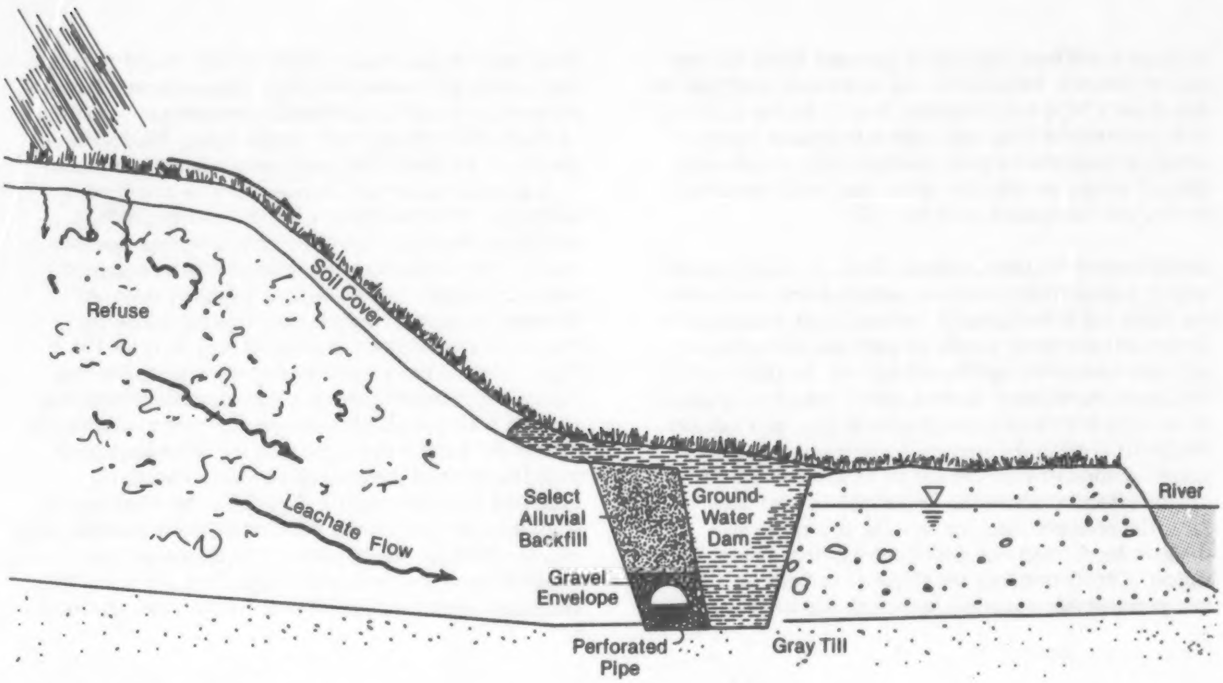

Figure 181. Site Cross Section[129]

purpose of the ground-water dam was to prohibit water originating in the adjacent river from flowing into the trench, which would have substantially increased the volume of wastewater.

Gradient Reversals. Controlling the movement of contaminated ground water by means of recharge and discharge wells has been practiced for several years. The major disadvantages include the commonly long pumping periods, well construction and maintenance costs, and the fact that the subsurface geology dictates system design. The cone of depression around a pumping well can be controlled by the discharge rate and thus change ground-water flow directions as well as velocity. Management of the cone or cones permits the operator to capture contaminants, which can then be diverted to a treatment plant. Well placement is particularly important since proper spacing and pumping rate are required to capture the contaminants. Moreover, well placement should be optimized so that as little uncontaminated water as possible is produced in order to reduce treatment costs.

Recharge wells are used to develop a hydraulic barrier. In this way they can be used to force the contaminant plume to move in preferred directions, such as toward a drain or discharging well.

The design of well systems is, in large part, based on trial and error methods coupled with experience. Herein also lies one of the most useful exercises of computer simulations because one can easily evaluate different schemes and estimate costs. Of course as in all simulations, the output can never be more valid than the input data.

Gradient reversal techniques are used at a great number of contaminated sites and nearly always play some role in containment methods, as is the case at the Rocky Mountain Arsenal.

Surface Sealing. Nearly all of the leachate that is generated from solid waste is derived by infiltrating rain and surface runoff that slowly percolate through the waste removing the water-soluble products. Clearly, one type of control to reduce the quantity of leachate generation is to stop or greatly reduce the volume of water that flows through the waste. One practical method is to provide a cover or cap that promotes surface runoff and prohibits infiltration. The cover, which would require continual maintenance and monitoring, could be constructed of clay, perhaps blended with bentonite or other additives, and used in combination with any of a number of liners sandwiched between clay layers.

Monitoring and maintenance of the cover are important to ensure that water cannot be impounded as the waste settles, that cracks caused by wetting and drying do not serve as a path through the cover, and that burrowing organisms do not penetrate the seal. Although cover monitoring and maintenance are commonly ignored after construction, the time and costs that they involve can be infinitely smaller than a cleanup operation. The substantial decrease in the size and concentration of a leachate plume brought about by surface sealing of the Windham, Connecticut, landfill is briefly discussed elsewhere in this report.

Source Removal. In some cases it is justifiable to remove a source of contamination and backfill the excavation with clean material. On the other hand, removal might be dangerous in itself, a secure site must be available to dispose of the excavated material, it will certainly be expensive, and the process might create additional water contamination problems because removal is likely to increase the rate of decomposition. As described elsewhere, once the unsaturated zone becomes contaminated, it might require years or even decades for the material to be slowly flushed from the zone and, throughout this time, it may continually or intermittently supply a leachate. Of particular concern are those sites where the source material extends below the water table or where the annual fluctuation of the water table causes cyclic wetting and drying.

Source removal is certainly a viable option, but it must be predicated on a sound technical and economic understanding of all of the potential consequences. Would it be more sound, for example, to provide an effective seal than to remove it? Or is it so toxic that there is no other choice? In many past restoration attempts removal was accomplished with little or no problem, other than cost.

Noncontainment

Sometimes a contamination problem is more apparent than real, but under emergency conditions or public/regulatory pressure little thought may be given to the consequences. As an example, let us assume there is a derailment and tank cars, containing 300,000 gallons of a special drilling fluid, are ruptured and the 30,000 mg/l chloride brine is spilled along the railroad embankment. At the end of two days all of the brine has infiltrated. The derailment occurred in the middle of the wide flood plain of a midwestern river, a river that flows generally no more than three or four months each year. The river channel in the vicinity of the spill lies along the north edge of the flood plain, as shown in Figure 182. In this rural area the nearest public water supply is at a recreational site a mile downgradient from the spill. What should be done?

Only limited hydrogeologic data are likely to be available, but many conditions can be inferred from a topographic map and a basic understanding of ground water geology. The few water wells in the flood plain indicate that the average thickness of the alluvium is about 50 feet. Drillers logs of the wells and an examination of borrow and sand pits indicate that most of the alluvium is a medium to coarse sand, which should have an effective porosity of about 20 percent. An estimate of ground-water velocity, 1.5 feet/day, is based on aquifer grain size and the gradient of the flood plain, which should be about the same as the water table. Under these conditions, an estimate of the longitudinal and transverse dispersivities should be about 70 and 15 ft. Since the major contaminant is chloride, it will not be retarded by the aquifer material, but rather will flow at the same velocity as the ground water.

If the above estimates are used as input for a simple computer transport model or the nomograph described earlier, one can obtain a general impression of the contaminant movement with time and then develop some guidelines for reclamation, if necessary. A simulation representing the contaminant plume one year after the spill is shown in Figure 183. Notice that the plume has migrated downgradient at least 1,500 feet and the largest

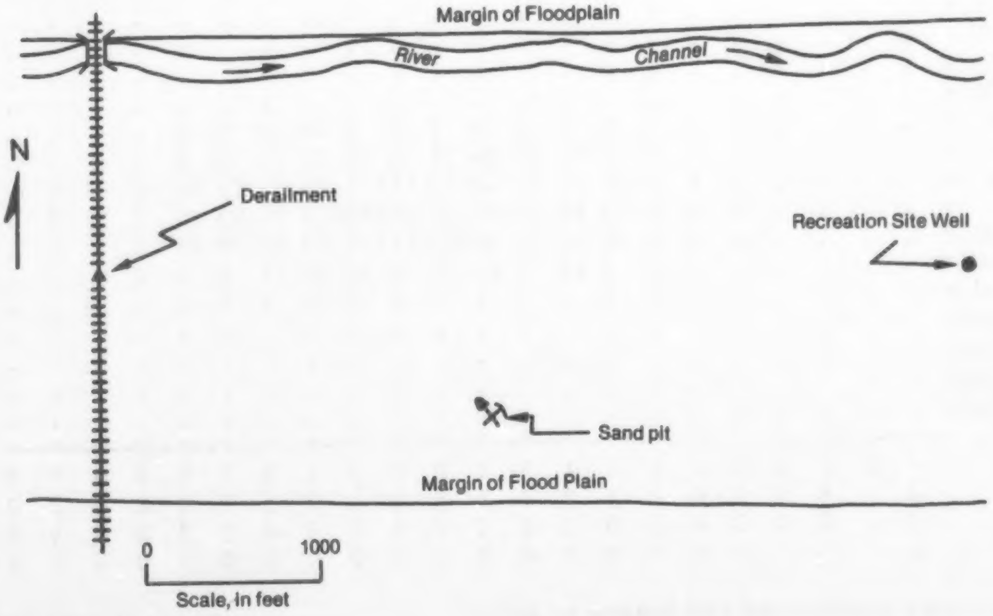

Figure 182. Map View of Railroad Spill Area

```
      PLUME AFTER 365 DAYS
      ------------------------------------------------------------------
1750 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
1500 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
1250 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
1000 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
750  +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
500  +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
250  +  2 17 68120 95 33  5  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
0    + 14  X496805627220 34  2  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-250 +  2 17 68120 95 33  5  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-500 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-750 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1000+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1250+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1500+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1750+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
      --+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+
        -  0  2  5  7  1  1  1  1  2  2  2  2  3  3  3  3  4  4  4  4  5
        2     5  0  5  0  2  5  7  0  2  5  7  0  2  5  7  0  2  5  7  0
        5     0  0  0  0  5  0  5  0  5  0  5  0  5  0  5  0  5  0  5  0
        0              0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
```

Figure 183. Chloride Distribution, in mg/l, 365 Days After the Spill

```
      PLUME AFTER 1825 DAYS
      ------------------------------------------------------------------
1750 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
1500 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
1250 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
1000 +  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
750  +  .  .  .  .  .  .  .  1  1  3  4  5  5  5  4  2  1  1  .  .  .  .
500  +  .  .  .  .  .  1  2  5 10 18 26 33 35 32 25 17  9  4  2  1  .  .
250  +  .  .  .  .  1  2  7 16 32 55 82103111101 79 52 29 14  6  2  1  .
0    +  .  .  .  .  1  4 10 23 47 81120151162148115 76 43 20  8  3  1  .
-250 +  .  .  .  .  1  2  7 16 32 55 82103111101 79 52 29 14  6  2  1  .
-500 +  .  .  .  .  .  1  2  5 10 18 26 33 35 32 25 17  9  4  2  1  .  .
-750 +  .  .  .  .  .  .  .  1  1  3  4  5  5  5  4  2  1  1  .  .  .  .
-1000+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1250+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1500+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
-1750+  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .  .
      --+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+--+
        -  0  2  5  7  1  1  1  1  2  2  2  2  3  3  3  3  4  4  4  4  5
        2     5  0  5  0  2  5  7  0  2  5  7  0  2  5  7  0  2  5  7  0
        5     0  0  0  0  5  0  5  0  5  0  5  0  5  0  5  0  5  0  5  0
        0              0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
```

Figure 184. Chloride Distribution, in mg/l, 1,825 Days After the Spill

concentration is 805 mg/l. If the river had been perennial, the plume would have moved toward it and eventually provided part of its flow. The concentration distribution of chloride in the plume 5 years after the spill is shown in Figure 184. The leading edge of the plume has moved nearly 5,000 feet from the spill but the maximum concentration is only about 162 mg/l.

This example illustrates that a few minutes with a simple computer model or nomograph clearly indicated that an accidental spill of the type described above really presented no problem at all because the aquifer diluted the contaminant to levels below drinking water standards. Furthermore, the limit on chloride in drinking water was established only for taste considerations. On the other hand, had the spill consisted of organic compounds the aquifer might not have been able to sufficiently dilute or retard the plume and corrective action might have been warranted.

In a great number of cases only one or two wells in the supply system are contaminated and these only with barely detectable concentrations. Depending on the health hazard of the contaminants, its concentration in the wells, and the volume pumped by the remaining wells, as well as on regulatory constraints, it might be possible to blend the water from all of the wells to reduce the concentration below detectable limits. There appears to be some philosophical differences to this solution among regulatory agencies. Some believe that if a contaminant is below detection limits that it would present no health hazard or at least an acceptable hazard. Others apparently have the opinion that if a part of the supply is known to be contaminated then it should not be used regardless of the concentration.

Some public water-supply systems contain excessive concentrations of ions that are derived from natural sources. These are largely inorganic substances that are likely to be more troublesome than hazardous, such as chloride or hardness. Excessive concentrations of these constituents might be the result of salt water intrusion or interaquifer leakage. In such cases the best control is probably a reduction of pumping, if possible, and blending with other supply sources that are less mineralized.

In places where the water supply system or the aquifer is grossly contaminated, it may be impossible to restore it despite the expenditure of great sums of money and time. Where this occurs, it might be necessary to simply abandon the system as a supply source. Monitoring and perhaps some reclamation would be required in order to reduce the potential for contamination of other sources of water.

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